ISSN 0082-3449

CONTRIBUTIONS IN MARINE SCIENCE

VOLUME 32 AUGUST 1991

CONTRIBUTIONS IN MARINE SCIENCE

VOLUME 32 AUGUST 1991

PUBLISHED BY

MARINE SCIENCE INSTITUTE
THE UNIVERSITY OF TEXAS AT AUSTIN
PORT ARANSAS, TEXAS 78373-1267

FOUNDED BY E. J. LUND IN 1945

This series will continue as
an irregular monographic series.

Editor: Patrick L. Parker
Editorial Assistant: Rebecca Ford
Technical Editor: Ruth Grundy

CONTRIBUTIONS IN MARINE SCIENCE (Formerly *Publications of the Institute of Marine Science*) is printed at irregular intervals by The Marine Science Institute and includes papers of basic or regional importance in marine science, with emphasis on the Gulf of Mexico and surrounding areas.

Issues are distributed at cost (no discounts possible), or on an exchange basis.

Reprints of individual articles are distributed through the authors only.

For orders (Volumes 7 onward plus four supplements are available), further information, or exchange arrangements please write to the Librarian, Marine Science Institute, The University of Texas at Austin, Port Aransas, Texas 78373-1267. This series will continue irregularly as a monographic series. Prospective contributors should contact the Editor, Contributions in Marine Science, Marine Science Institute, The University of Texas at Austin, Port Aransas, Texas 78373-1267 for further information.

Copyright © 1991

DEDICATED TO DONALD E. WOHLSCHLAG, EDITOR 1975–1990

This volume of Contributions in Marine Science is dedicated to Professor Donald Eugene Wohlschlag. It is done so in appreciation for the years of service he has given the University as teacher, Director and, as many of our readers know, as Editor of *Contributions in Marine Science* from 1975 to 1990.

Curly, as Professor Wohlschlag was nicknamed long ago, came to The University of Texas as Director of the Marine Science Institute and Professor of Zoology in September 1965. From Stanford to Texas was a long drive and a big change for Curly and Marjorie. But they quickly settled into new duties. Marjorie organized and ran the Port Aransas Independent School District's kindergarten, perhaps the first free one in Texas and Curly took up the duties of the Director. Geraldine Ard, Curly's secretary, loved to tell the story of his first dictated letter. It began "Dear Yo-Yo." Some careful investigation revealed that Yo-Yo was Paul Dayton, one of Curly's early field students at Stanford.

During Curly's term as Director (1965-1970), the Board of Regents authorized a major building program at the Institute and through his efforts, the University acquired 48.78 acres of beach front property from Nueces County and the Federal Government. This happened about the time of Hurricane Beulah in 1967. We never did know whether Curly used the actual hurricane as an argument for a new building or whether he just threatened the Regents with a hurricane if they did not build a new lab. Either way, a generation of staff and students benefited from his leadership.

Many of these students were Curly's own. During his career at Texas he has supervised 8 Ph.D. and 8 M.A students. These men and women continue the research and problem solving that Curly started them on at positions across the country. Curly was devoted to hands on experience for his students and technicians. It became a key part of their training and has served them well. Of course, there is some risk in letting young folks find their own way. Now we can laugh at the young man who needed a little fiber glass resin and poured his leftover resin back into the five gallon can. It made a great boat anchor.

Curly is one of the few scientists who has done research in both the Arctic and the Antarctic. He started in the Arctic in the early 1950s during his Stanford days. No one knows why he switched; adventure, romance, money, science? It must have been opportunity. In the year prior to leaving Stanford for UT, Curly was running the biological laboratories in the Antarctic for the National Science Foundation. His years in those cold places may explain his move to Texas. Texas has not been dull for him either. Besides the heat and administrative nightmares of running a facility remote from the campus, he still holds the record for having the roof of his home blown off the most times.

The best part of the Donald E. Wohlschlag story is that he still is a colleague and a friend. As an emeritus Professor, he comes in every day when he is not traveling. His counsel and support for the marine program over some stormy years mean a lot to all us. We are glad that Curly and Marjorie came our way.

TABLE OF CONTENTS

Dedication to D.E. Wohlschlag iii

Stable isotopic analysis of organic carbon in water and sediment from the Houston Ship Channel, Texas, USA
Michael E. Caughey 1

No. 2 fuel oil effects on *Spartina alterniflora* in a Texas salt marsh **James W. Webb and Steve K. Alexander** 9

A new record of *Paulinella ovalis*: Filosea: Euglyphina
Susan E. Safford 21

Factors influencing the standing crop of diatom epiphytes of the seagrass *Halodule wrightii* Aschers. in South Texas seagrass beds **Jerilyn Jewett-Smith** 27

Effects of the number of short shoots and presence of the rhizome apical meristem on the survival and growth of transplanted seagrass *Thalassia testudinum*
David A. Tomasko, Clinton J. Dawes and Margaret O. Hall 41

The effect of freshwater inflow on macrobenthos in the Lavaca River delta and upper Lavaca Bay, Texas
Richard D. Kalke and Paul A. Montagna 49

A laminar-flow choice chamber for testing the responses of postlarval penaeids to olfactants
Mark C. Benfield and David V. Aldrich 73

Long-term trends in the abundance of the copepod *Acartia tonsa* in the Calcasieu Estuary **Michael Vecchione** 89

Ichthyoplankton off west Louisiana in winter 1981-1982 and its relationship with zooplankton biomass
James H. Cowan, Jr. and Richard F. Shaw 103

Occurrence of fat snook (*Centropomus parallelus*) in Texas: evidence for a range extension **J.H. Martin and T.L. King** 123

Seasonal variation in relative abundance and species diversity of fishes in South Bay **John H. Hook** 127

Contributions in Marine Science. (1991)Vol. 32.

STABLE ISOTOPIC ANALYSIS OF ORGANIC CARBON IN WATER AND SEDIMENT FROM THE HOUSTON SHIP CHANNEL, TEXAS, USA

Michael E. Caughey[1]

University of Texas at Austin
Marine Science Institute
Port Aransas, Texas 78373-1267

ABSTRACT

In July 1986 water and sediment samples were collected at 14 stations along the Houston Ship Channel. The stable isotope ratios ($^{13}C/^{12}C$) of dissolved, particulate and sediment organic carbon are useful in estimating the degree and extent of petrochemical pollution. Comparisons of the 1986 water sample data with those from the same stations 18 years earlier support other evidence that water quality in the Houston Ship Channel has improved. While the mean $\delta^{13}C$ value for POC was virtually unchanged, the mean $\delta^{13}C$ value for DOC increased from -32 per mil in 1968 to -27 per mil in 1986, as would be expected if petrochemical pollution had decreased. The weight percent organic carbon, the C:N ratio and the isotopic evidence of petrochemicals in the sediment decreased as distance along the channel from the inland turning basin increased. In 5 sediment samples, between 20 and 60 weight percent of the organic carbon was extractable using methanol-toluene; the $\delta^{13}C$ values of the extracted organic matter fall between -28 and -26 per mil.

INTRODUCTION

The Houston Ship Channel (HSC) is an 80 kilometer-long waterway (Fig. 1)

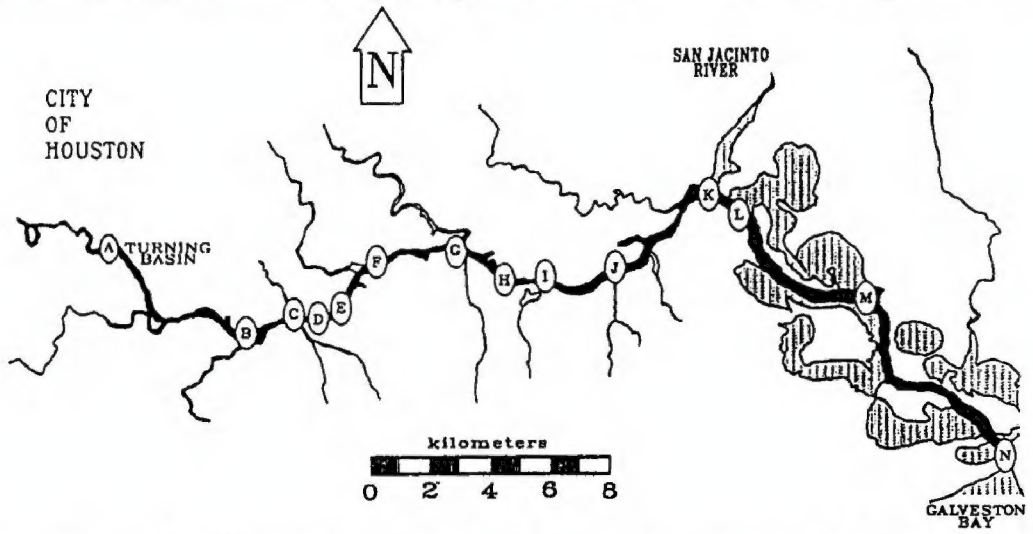


FIG. 1. Water and sediment sample locations on the inland portion of the Houston Ship Channel, Texas, USA.

---

[1] Present address: Illinois State Water Survey, Office of Environmental Chemistry, 2204 Griffith Drive, Champaign, Illinois, 61820-7495.

which begins at Bolivar Roads in the Gulf of Mexico, crosses Galveston Bay and continues inland following portions of the San Jacinto River and Buffalo Bayou to a ship turning basin near metropolitan Houston, Texas. Approximately one half of U.S. chemical manufacturing and one third of the U.S. petroleum industry are located in the Houston-Galveston Bay area. Water quality along the HSC has been notoriously low; in the 1960's the EPA named it among the ten most polluted water bodies in the United States. At that time industry was responsible for approximately 70 percent of the total organic carbon input to the HSC. As result of major clean-up effort, by 1980 industrial effluents reportedly accounted for only 20 percent of the total input (Blodgett, Smerdon, McGarity and Stewart 1986).

Calder and Parker (1968) demonstrated that the $\delta^{13}C$ values of the organic compounds manufactured from natural gas (see Equation 1) are sufficiently

EQUATION 1

$$\delta^{13}C = \frac{(^{13}C/^{12}C)\ \text{sample} - (^{13}C/^{12}C)\ \text{std}}{(^{13}C/^{12}C)\ \text{std}} \times 1000$$

different from those of other types of dissolved organic carbon (DOC) that such petrochemical pollution can be traced in aquatic systems without resorting to involved chemical separations. They measured $\delta^{13}C$ values as negative as -50 per mil for Houston Ship Channel DOC. To determine whether carbon stable isotope ratio analyses would confirm reports that the percentage of industrial organic input has decreased markedly, another set of samples was collected in July 1986 at 13 of the same locations along the HSC.

In addition to water, grab samples of sediment were also taken at each station. Channel depth varies from 11 m in the upper reaches to 14 m in the lower; a proposal to deepen it to 15 m and widen it to a minimum of 137 m is currently being considered. The nature and concentration of sediment organic components which might be returned to the water column during the dredging process, as well as those which could be eventually leached from the dredge spoil, are major factors in evaluating the environmental impact of such a project. Stable isotope ratio measurement provides a rapid preliminary assessment of the sources of the organic carbon contained in HSC sediments.

METHODS

Water samples were collected from the approximate center of the channel just below the surface using a Niskin bottle. Duplicate samples were stored frozen until analysis. Particulate organic carbon (POC) was collected on pre-combusted Whatman GF/C filters. Dissolved organic carbon was converted to carbon dioxide for mass spectrometry using peroxydisulfate wet oxidation (Calder and Parker 1968) with the following two modifications.

1) Interfering chlorine compounds, formed by the oxidation of chloride in the samples, were removed from the gas mixture by recirculation through copper turnings heated to 400°C.

2) Teflon stopcocks were used in place of the glass break seals previously used on the reaction bulbs.

The precision of replicate $\delta^{13}C$ values for dissolved carbon standards was better than ± 0.4 per mil, while sample replicates agreed within ± 0.5 per mil. It should be noted that the nitrogen sparging used to remove the inorganic carbon following sample acidification also removes volatile organic carbon compounds from the water samples.

Sediment from shallow (<30 cm) grab samples was collected at the same locations as the water samples. When the sediment contained plant material, the pieces large enough to grasp with forceps were removed before the samples were frozen. Inorganic carbon and acid-labile sulfides were removed from thawed sediment subsamples by adding excess 1 M hydrochloric acid. In order to retain the bulk of the non-volatile organic carbon in the sediment, the reaction mixture was not filtered and the excess acid was allowed to evaporate in a fume hood at room temperature. The dried sediment was passed through an NBS number 40 sieve prior to analysis. Weighed subsamples (5 to 10 g) of dry, acid-treated sediment from 5 of the 14 stations were extracted by stirring them for 18 hrs at room temperature in 250 ml of 1:2 (V:V) methanol-toluene. The slurries were then filtered through glass fiber, followed by two additional methanol-toluene washes of the residual sediment. The solvent was removed from the extracts and from the extracted sediment by prolonged vacuum evaporation. Since only non-volatile carbon compounds were retained in the samples analyzed, the measured values probably underestimate the amount of petrochemical pollution in the sediments.

The carbon dioxide produced via sealed-tube, cupric oxide oxidation (Sofer 1980) from samples of POC, whole sediments, sediment extracts and sediment residues was measured using a calibrated manometer. Isotope ratio mass spectrometry was done using a VG Micromass model 602E. Precision for this method is better than ± 0.2 per mil. Atomic ratios of carbon to nitrogen were determined using a Perkin-Elmer 240B elemental analyzer.

RESULTS AND DISCUSSION

The results of the DOC and POC analyses are summarized in Table 1. To facilitate intercomparison, the results from the 1968 study are included with the $\delta^{13}C$ value recalculated relative to the Peedee Belemnite (PDB) isotopic standard, rather than to the NBS-20 standard as originally reported by Calder and Parker (1968). The $\delta^{13}C$ values for natural background marine organic matter range from -20 to -15 per mil, while values for natural terrigenous organic matter typically fall between -27 and -20 per mil. Organic matter with $\delta^{13}C$ values more negative than -27 per mil probably contains appreciable amounts of carbon derived from natural gas or petroleum (Degens 1969).

Though it is difficult to find a true analog to serve as an unpolluted control, the Lavaca River and Lavaca Bay (located approximately 130 km south of Galveston Bay) represent a much less industrialized river-estuary system. A typical $\delta^{13}C$ value for Lavaca River DOC falls in the range of -24 to -20 per mil, while POC there varies between -26 and -22 per mil (Jones 1986).

As industries contribute progressively smaller percentages of the total DOC in the HSC, the population in the surrounding area continues to grow. Consequently, municipal wastewater treatment plants are becoming increasingly important as sources of DOC. Reimers (1968) measured $\delta^{13}C$ values for the effluents of ten wastewater treatment plants in south-central Texas. The DOC $\delta^{13}C$ values ranged from -25 to -21 with a mean of -24 per mil, while values for POC were from -23 to -19 with a mean of -21 per mil. This means that domestic effluents can be distinguished on the basis of $\delta^{13}C$ values

TABLE 1
Results of dissolved and particulate organic carbon stable isotope ratio analyses ($\delta^{13}C$ values are permil *vs.* PDB)

Station	km from turning basin	$\delta^{13}C$ DOC 1968	$\delta^{13}C$ DOC 1986	$\delta^{13}C$ POC 1968	$\delta^{13}C$ POC 1986
A	0.0	-32.3	-27.1	-22.4	-23.7
B	6.8	-32.6	-25.1	-25.3	-26.5
C	8.6	-30.4	-29.6	-24.2	-25.3
D	9.7	-31.2	-25.5	-27.4	-24.2
E	10.2	-28.0	-31.0	-25.5	-25.3
F	12.0	nd.***	-28.6	-24.4	-25.9
G	14.9	-49.9	-27.1	-28.5	-25.0
H	16.7	nd.	-25.1	nd.	-24.4
I	19.1	-30.1	-25.5	-25.8	-24.0
J	21.0	-28.2	-27.6	-27.0	-26.4
K	25.1	-28.6	-26.8	-25.1	-26.5
L	27.1	-29.3	-26.0	-27.1	-26.4
M	32.7	-33.1	-27.4	-26.9	-26.6
N	40.1	nd.	-28.3	-24.1	-26.7
mean		**-32.1**	**-27.2**	**-25.7**	**-25.5**

*** ND = not determined

from industrial effluents containing petrochemicals, but they cannot be distinguished unambiguously from natural background organic matter.

The mean of the $\delta^{13}C$ values for HSC DOC in 1968 was -32 per mil, clearly indicating petrochemical contamination. The 1986 DOC mean for $\delta^{13}C$ was -27 per mil, which is much nearer the range of values typical of naturally occurring terrigenous organic matter. The two sample sets are statistically different according to the paired t-test ($p \geq 0.05$). Assuming that the mean $\delta^{13}C$ value for petrochemical pollution is -36 per mil, the mixing equation discussed by Calder and Parker (1968) suggested that the 5 per mil change in the mean $\delta^{13}C$ value between 1968 and 1986 would correspond to approximately a 50% reduction in the fraction of DOC attributable to petrochemical effluents. While the range of $\delta^{13}C$ values for POC was slightly wider in 1968 than in 1986, the means were virtually identical at -26 per mil. Either the average composition of the POC was more or less the same in 1986 as in 1968, or else carbon stable isotope ratio measurements are insensitive to any change which took place.

A major factor in comparing water quality for the two years should certainly be the concentration of DOC. Unfortunately a tropical storm occurred three

days prior to the 1986 sampling cruise, presumably causing dilution throughout the channel. Evidence of this is seen in the low surface water salinity values, which range from 2 ‰ in the turning basin to 4 ‰ in Galveston Bay. In comparison, Hann (1970) measured surface salinities near station "N" (see Fig. 1) in the summer of 1966 which ranged from 1 to 15 ‰, with a mean of approximately 7 ‰. In 1968 the DOC concentrations ranged from 3 to 26 $mg \cdot L^{-1}$ with a mean of 10 $mg \cdot L^{-1}$; 1986 values ranged from 2 to 12 $mg \cdot L^{-1}$ with a mean of 5 $mg \cdot L^{-1}$. POC concentrations in 1968 ranged from 2 to 19 $mg \cdot L^{-1}$ with a mean of 7 $mg \cdot L^{-1}$; in 1986 they ranged between 2 and 5 $mg \cdot L^{-1}$ with a mean of 3 $mg \cdot L^{-1}$, again suggesting dilution by rainfall to roughly one half the presumed normal concentration.

The results of the acid-treated sediment analyses display well defined trends relative to location along the length of the channel. In general, the C:N ratios decrease, the weight percentages of total organic carbon (TOC) decrease, and the $\delta^{13}C$ values of the total sediment organic matter become less similar to those typical of petrochemicals as distance from the turning basin increases (Fig. 2 and Table 2). Seven of the 1986 stations are near locations where Smith (1968) measured the chemical oxygen demand (COD) of sediment samples. Because of significant differences in the sampling methods and in the completeness of the two types of oxidation, the values from the two studies cannot be compared directly. However, the ratios of the 1968 COD values to the 1986 TOC values at similar locations are constant to within a factor of two. This rough comparison shows that the distribution of sediment organic matter along the length of the channel did not change dramatically over a period of nearly 20 years.

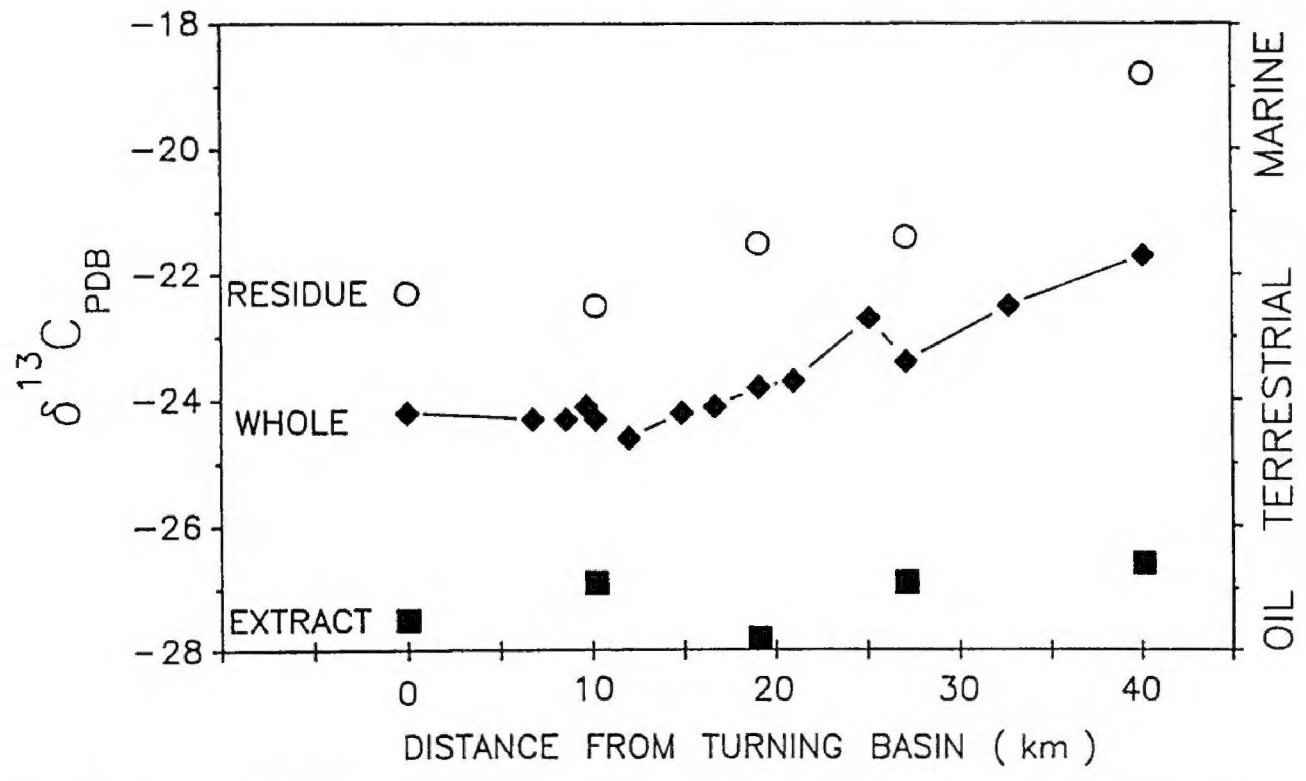


FIG. 2. $\delta^{13}C$ values of whole sediments, extracted sediments and lipid extracts showing trends with distance along the HSC.

TABLE 2
Results of analyses of sediments from the Houston Ship Channel. (Extractions were done with 1:2 (V:V) CH_5OH-Toluene.)

	whole sediment			extracted sediment		extract	
STN	WT % Ogr. C	C : N (atom)	$\delta^{13}C$ permil	WT % Org. C.	$\delta^{13}C$ permil	Fractn. of TOC	$\delta^{13}C$ permil
A	2.7	12.1	-24.2	1.6	-22.3	0.42	-27.5
B	2.8	12.8	-24.3				
C	2.5	12.9	-24.3				
D	2.6	11.1	-24.1				
E	2.6	10.1	-24.3	1.9	-22.5	0.23	-26.9
F	2.7	9.9	-24.6				
G	2.5	9.9	-24.2				
H	1.5	9.3	-24.1				
I	1.8	9.3	-23.8	1.2	-21.5	0.33	-27.8
J	1.8	8.6	-23.7				
K	1.7	7.0	-22.7				
L	1.4	7.2	-23.4	0.9	-21.4	0.36	-26.9
M	1.2	6.5	-22.5				
N	1.2	5.7	-21.7	0.4	-18.8	0.58	-26.6

The trends mentioned above reflect three types of progressive change in the composition of the sediment organic matter. The first is the usual decrease in the C:N ratio observed when moving from a terrigenous-dominated to a marine-dominated environment. Of course, addition of carbon-rich pollutant to the natural sediment organic matter will also increase the C:N ratio. The second trend could be attributed to the combined effects of:

1) greater input of natural and/or pollutant organic matter in the inland portions of the channel and
2) enhanced organic matter preservation where dissolved oxygen levels and sediment resuspension rates are lower.

Thirdly, the change in $\delta^{13}C$ values is likewise consistent, both with the natural transition from terrigenous to marine organic matter, and with a decrease in the petrochemical pollutant content of the sediment. Of course departures from these trends occur, such as the TOC, C:N and $\delta^{13}C$ values at station "K". The inflow of the San Jacinto River at the point appears to introduce additional organic matter of a type which is richer both in nitrogen and in $\delta^{13}C$ than the generalized trend lines would otherwise predict.

One way to attempt to distinguish between the natural and pollutant carbon in sediment is to do a simple lipid extraction. Figure 2 and Table 2 show the results of such analyses. Although the $\delta^{13}C$ values of the whole sediment (shown by diamonds) are within the normal range for terrestrial organic

matter, lipid extraction shows that it is probably a mixture of natural and anthropogenic components. The non-extractable carbon in the HSC sediments (shown by circles) appears to be predominantly a gradational mixture of natural marine and terrigenous organic matter. The less negative $\delta^{13}C$ values for the non-lipid residue at stations nearer Galveston Bay indicate increasing percent contributions of marine organic matter to the total organic carbon. The $\delta^{13}C$ values of sediment organic matter extracted by methanol-toluene (shown by squares) are characteristic of petroleum and petrochemicals. That extractable, non-volatile organic carbon accounts for between 20 and 60% of the TOC. In areas where sediments are not greatly affected by industry or shipping, typically less than 7% of the TOC is extractable in this way (Parker 1967). In other words, HSC sediments are contaminated with an unknown mixture of organic compounds to a degree which makes it advisable that they be chemically characterized before dredging begins.

Summary: The results of stable carbon isotope ratio measurements are consistent with reports that industrial effluents contributed a markedly smaller fraction of the DOC in 1986 than they did in 1968. The isotopic composition of the POC was not found to have changed significantly. Sediments from the channel bottom contain as much as 1 weight percent of extractable, petroleum-like organic matter of unknown chemical composition.

ACKNOWLEDGMENTS

The author would like to thank Robert Jones, Director UTMSI, for authorizing this as the maiden sampling cruise of the newly refitted R/V Longhorn. Thanks are also owed to Richard Anderson, Pat Parker and the Longhorn Crew for their assistance in sample collection, and to Richard Scalan and Terry Whitledge for helpful editorial comments. Portions of this work were presented as part of the Symposium on Effects of Off-Shore and On-Shore Industrial Activities in the Gulf of Mexico at the American Chemical Society Meeting at New Orleans in August, 1987. The University of Texas at Austin Marine Science Institute Contribution No. 820.

LITERATURE CITED

BLODGETT, T., E. SMERDON, T. McGARITY and R. STEWART. 1986. An institutional and legal assessment of a instream aeration project in the Houston Ship Channel, Technical Report No. 220, Center for Research in Water Resources, University of Texas at Austin.

CALDER, J.A. and P.L. PARKER. 1968. Stable carbon isotope ratios as indices of petrochemical pollution of aquatic systems. *Environmental Science & Technology*. 2:535-539.

DEGENS, E.T. 1969. Biogeochemistry of stable carbon isotopes. pp. 304-329. In G. Eglinton and M.T.J. Murphy (eds), *Organic Geochemistry*.

HANN, R.W. 1970. Analysis techniques for the Houston Ship Channel. *Journal of the Waterways and Harbors Divisions*. 96:171-180.

JONES, R.S. 1986. Studies of freshwater inflow effects on the Lavaca River delta and Lavaca Bay, Texas. University of Texas at Austin Marine Science Institute Technical Report No. TR/86-006.

PARKER, P.L. 1967. Fatty acids in recent sediment. *Contributions in Marine Science.* **12**:135-142.

REIMERS, R.S. 1968. A stable carbon isotopic study of a marine bay and a domestic waste treatment plant. Thesis, University of Texas at Austin, Texas.

SMITH, R.H. 1968. Survey of organic content in bottom sediments of the Houston Ship Channel. Thesis, Texas A&M, College Station, Texas.

SOFER, S. 1980. Preparation of carbon dioxide for stable carbon isotope analysis of petroleum fractions. *Analytical Chemistry.* **52**:1389-1391.

Contributions in Marine Science. (1991) Vol. 32.

NO. 2 FUEL OIL EFFECTS ON *SPARTINA ALTERNIFLORA* IN A TEXAS SALT MARSH

James W. Webb and Steve K. Alexander[1]

Marine Biology Department
Texas A&M University at Galveston
Galveston, Texas 77553-1675

ABSTRACT

The effects of No. 2 fuel oil on *Spartina alterniflora* and the recovery potential of affected plants were investigated in a Galveston Bay, Texas salt marsh. To simulate various oil spill conditions, No. 2 fuel oil was applied to 4-m^2 plots of *S. alterniflora* as four treatments (control, sediment only, sediment plus lower plant surfaces, and sediment plus entire plant surfaces). Live stem density was compared among treatments at monthly intervals for one year to determine initial damage to plants and subsequent recovery. Live stem densities in sub-plots established in the middle and on the outer edge of plots were compared to determine whether plant recovery occurred from surviving plants or rhizome growth into plots from the surrounding marsh. Initial damage to plants increased and regrowth from rootstock decreased as oil coverage of above-ground foliage increased. With complete coverage of foliage by oil, above-ground plant material died rapidly and regrowth from rootstock did not occur. When oil was applied to the sediment alone, stem density was slightly reduced, presumably by penetration of oil so that root contact occurred. Low amounts of sediment hydrocarbons plus new stems produced from rhizomes growing into plots from the surrounding marsh indicated that there was little residual oil toxicity in sediments. Hydrocarbons in the root system after oil application to foliage indicated that oil components were translocated from foliage to roots and rhizomes. Plant regeneration from roots may have been affected.

INTRODUCTION

Coastal marshes have many values. They produce large amounts of plant biomass, which provides energy to the estuarine ecosystem in the form of detritus (Odum, Zieman and Heald 1972). Marshes are instrumental in prevention of shoreline erosion (Knutson, Ford and Inskeep 1981). They are an important habitat for birds, mammals, and estuarine organisms (Davidson and Chabreck 1983, Beccasio, Fotheringham, Redfield, Frew, Levitan, Smith and Woodrow 1982). Approximately 66% of the commercially important fish and shellfish of the Atlantic and Gulf waters of the United States utilize estuarine wetlands during young and juvenile stages of their life cycle (McHugh 1968; Lindall and Saloman 1977; Peters, Ahrenholz and Rice 1978; Herke 1971; Mock 1966; Conner and Truesdale 1972; Montague, Bunker, Haines, Pace and Wetzel 1981). Some estuarine species invade the marsh,

[1] Present address: Department of Biology, University of Mary Hardin-Baylor, Belton, Texas 76513.

particularly the *S. alterniflora* zone, at flood tide in significant numbers to derive nutrition and protection from predators (Zimmerman and Minello 1984a). Brown shrimp (*Penaeus aztecus*) strongly select for vegetated habitat dominated by *S. alterniflora* (Zimmerman and Minello 1984b; Zimmerman, Minello and Zamora 1984).

Because of its intertidal location in coastal marshes, *S. alterniflora* is often affected by oil spills, including spills of No. 2 fuel oil. For example, the west Falmouth spill in 1969 (Burns and Teal 1971) and the Bouchard barge spill in 1974 (Hampson and Moul 1978) in Buzzards Bay, Massachusetts caused serious damage to salt marshes. In February 1985, a leakage of 1.3 million gallons from a pipeline impacted salt marshes at Port Arthur, Texas (George Guillen, Texas Parks and Wildlife, personal communication). Recently, a 400,000 gallon No. 2 spill on Long Island, New York escaped into salt marshes following a tank valve failure (Christopher Gross, Long Island Lighting Company, personal communication).

Crude oils and heavy fuel oils may have a negligible impact on salt marshes (Hershner and Moore 1977; Delaune, Patrick and Buresh 1979; Crow 1974). Death of above ground plant parts may occur when crude and heavy fuel oils cover stems and leaves of *S. alterniflora*; however, regeneration of stems from roots and rhizomes usually results in regrowth to a level equal to pre-oil spill conditions within several months to a year (Webb, Tanner and Koerth 1981; Webb, Alexander and Winters 1985; Alexander and Webb 1985). Complete death of large areas of *S. alterniflora* generally occurs only when high levels of these oils accumulate in the sediment (Holt, Rabalais, Rabalais, Cornelius and Holland 1978; Alexander and Webb 1987; Krebs and Tanner 1981).

The effect of No. 2 fuel oil on *S. alterniflora* is generally faster, more severe, and long-term than for crude or heavy fuel oils. Straw colored leaves and immediate death of plants are frequently observed (Webb *et al.* 1985; Alexander and Webb 1985; Hampson and Moul 1978; Sanders, Grassle, Hampson, Morse and Price 1977). In several studies, death resulted in lack of recolonization and eventually erosion of the salt marsh (Hampson and Moul 1978; Hershner and Lake 1980).

The possibility that death of plants could result in long-term effects on the marsh warrants further investigation, especially the degree of damage and likelihood of recovery of *S. alterniflora*. This information is vital to determine appropriate actions following a spill which enters a salt marsh.

There appear to be several mechanisms by which *S. alterniflora* can be damaged and fail to recover when No. 2 fuel oil enters a *S. alterniflora* community. One possibility is that No. 2 fuel oil may act like a contact foliage active herbicide as described by Scifres (1980). Contact herbicides kill only that plant tissue directly contacted by the herbicide. Damage may be so severe and widespread upon contact with foliage that plants are not able to regenerate new growth. In the case of oils, greatest acute damage is generally caused by very toxic, oxidized oils and aromatics, which stop respiration and cause widespread injury and death (Baker 1970).

A second possibility is that root systems may be damaged by translocation of toxic components from leaves to below-ground biomass following foliage contact with No. 2 fuel oil. Translocation occurs as hydrocarbon components penetrate the leaves and stems of plants and move through the circulation system to roots and other organs distant from the point of entry. As a result, long-term damage occurs because of impairment of reproduction by the roots and rhizomes.

Herbicides and possibly No. 2 fuel oil can act differently with environmental conditions and with the amount of toxic components in contact with plant foliage. For example, excessive amounts of a translocation type herbicide can cause such severe damage to tissue at the point of contact that translocation of the herbicide does not occur because of impairment of physiological functions (Scifres 1980). As a result damage remains localized.

A third possibility is that high concentrations of toxic components may enter sediments and may affect the root system, cause death, and prevent plants from recolonizing. The death of plants due to residual toxicity in the sediment has been described by Teal, Burns and Farrington (1978). The type soil can be a factor in oil penetration, however. Oil may not penetrate and accumulate in predominantly mineral soils, particularly clay (Webb, *et al.* 1985), as readily as in organic soils in which long-term residual effects have been reported (Hampson and Moul 1978; Burns and Teal 1971).

This study was designed to further document the effects of No. 2 fuel oil on *S. alterniflora* and determine the recovery potential of *S. alterniflora* in a Gulf Coast salt marsh. To accomplish our objectives experimental plots were treated with No. 2 fuel oil at different amounts of plant coverage to simulate the oil coverage that would occur with various tide situations. Oil floating into marshes on spring or wind pushed tides tend to coat entire plants as the tide ebbs. During neap tide conditions (lower range between high and low tides) oil generally coats plants only part way up stems or only covers the sediment surfaces. In our study live stem counts were made periodically to determine initial effects and to monitor recovery. Sub-plots were established to evaluate whether regrowth occurred and whether it started from surviving treated plants or from rhizome growth into plots from the surrounding marsh. Sediment analyses were made to determine initial soil penetration by the oil and residual oil in the sediment.

METHODS

Sixteen 4-m^2 plots were established 1 m apart parallel to the shoreline (20 m inland) in a Galveston Bay, Texas salt marsh. Four treatments were randomly assigned in each of four replications of a randomized complete block design. The four treatments of No. 2 fuel oil were: no oil, 1.0 L per m^2 applied to the sediment surface, 1.5 L per m^2 applied to sediment and lower foliage of *S. alterniflora*, and 2.0 L per m^2 applied to the sediment surface and entire foliage of *S. alterniflora*. Oil was applied to plots with garden sprayers at low tide on September 2, 1983.

In a nearby marsh area on May 3, 1984, four 1-m^2 plots of *S. alterniflora* were sprayed with 2 L per m^2 of No. 2 fuel oil so that the entire plant foliage was covered with oil. Care was taken not to

spray the sediment, but some oil did reach the sediment. Four days later roots and rhizomes in the treated plots and 4 untreated plots were removed, generally to a depth of 20 cm, and washed free of soil. Roots and rhizomes were macerated in methylene chloride. Hydrocarbons were extracted with methylene chloride. Extracted plant material was dried and weighed. The oil extract was filtered on a Whatman GF/C glass fiber filter and concentrated on a rotary evaporator. The remaining solvent was evaporated in a tared evaporating vial at room temperature. The weight of oil residue was determined by increase in vial weight and then converted to a dry plant weight basis.

Initial damage to above-ground portions of *S. alterniflora* and oil movement was visually observed at 1 and 3 days after oil application. At monthly intervals, live stems, which included all stems with any green color, were counted and compared in 4 inner (I) and 4 outer (O) 0.25-m^2 sub-plots of all plots (Fig. 1) to determine whether regrowth occurred from surviving treated

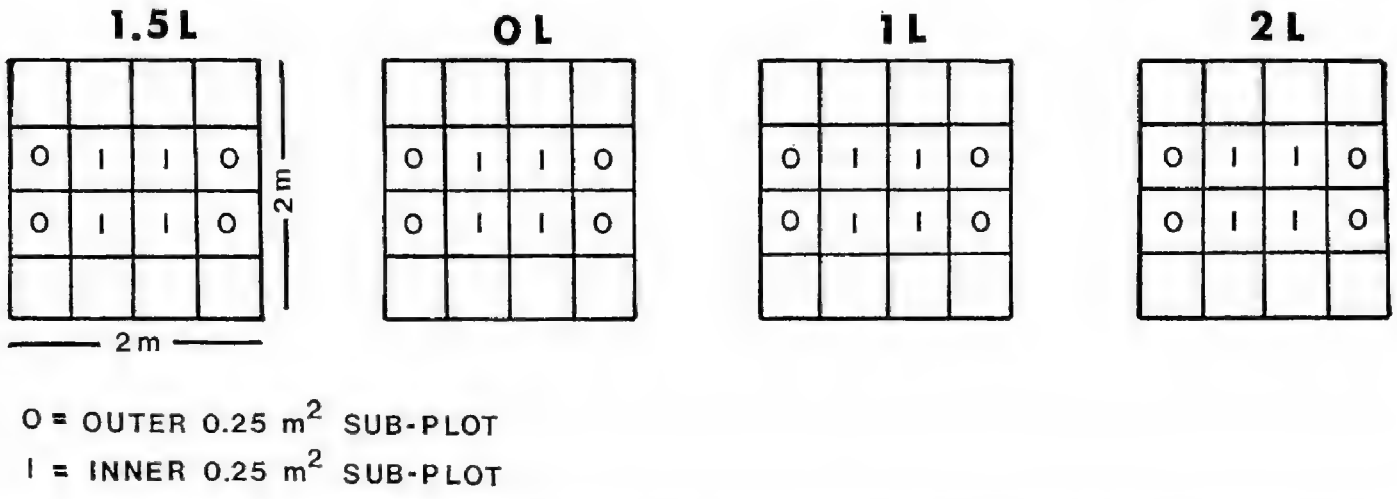


FIG. 1. Plot design showing inner and outer sub-plots for one replication of oil treatment study.

plants or from rhizomes penetrating the plots from non-treated areas (greater stem density in outer sub-plots). Equal numbers between outer and inner plots indicated that either no initial damage occurred from oil treatment or that recovery occurred from oiled plants. The sub-plots along the seaward and landward edges of the line of plots were eliminated from enumeration to avoid sub-plots possibly affected by trampling. Counts were discontinued after one year when it became apparent that growth of rhizomes from the surrounding marsh had reached inner plots of the heaviest treatment and new stems were emerging from those rhizomes. Numbers of stems in oil treated plots (pooled inner and outer sub-plots) were compared statistically to control plots (no oil) and to other treatment levels to indicate damage to above-ground growth and plant recovery after initial damage.

For each 4-m^2 plot three sediment cores (to a depth of 5 cm) were taken at 10 days and 2 months after oil application. Cores were pooled for each plot, frozen, and analyzed later for total hydrocarbons. The oil in 100 g of wet sediment was extracted with three successive 75 ml volumes of methylene chloride and oil residue was determined similarly to plant material described above. Oil residue was converted to a dry sediment weight basis.

Plant and soil data were evaluated by analysis of variance procedures and Duncan's Multiple Range Test, utilizing computerized Statistical Analysis System (SAS Institute, Inc. 1985).

RESULTS

One day after application with No. 2 fuel oil, plots with entire foliage coverage with oil were noticeably different from other plots. Leaves and

upper stems of *S. alterniflora* had turned from a green to a straw color, but stem bases were still green. Plants in all other plots looked normal. There was little evidence of oil in treated plots except for an oil sheen on the sediment surface. Oil was seen in adjacent marsh areas, indicating that high tides dispersed the oil.

Three days after application, plants in the plots with entire coverage still had green coloration only on stem bases. In plots with lower stem coverage, many plants had a slight yellowish coloration while plants in plots with oil applied to sediment only visually appeared unaffected by the oil. Live stem density of *S. alterniflora* in each oil treatment was significantly different from the control and each other treatment at each sampling period through one year (Table 1). The number of live stems decreased with increasing oil

TABLE 1

Mean number[1] of live stems in inner and outer 0.25-m^2 sub-plots for each level of treatment (no oil - control, sediment only treated at 1 L of oil per m^2, lower half of plants sprayed at 1.5 L per m^2, entire plants covered at 2 L per m^2) for 4 m^2 plots sprayed with No. 2 fuel oil on September 2, 1983.

	TREATEMENTS							
	Control		Sediment		Lower stems		Entire plant	
	0 l/m^2		1 l/m^2		1.5 l/m^2		2.0 l/m^2	
Date[2]	Inner	Outer	Inner	Outer	Inner	Outer	Inner	Outer
Sep 29	34.7	33.8	19.5	21.3	13.1	14.9	0.0	0.0
Oct 26	47.7	49.1	26.6	46.4	11.8	13.5	0.1	0.3
Jan 04	38.6	39.8	16.5	23.1	8.9	7.4	0.3	1.8
Feb 01	31.4	27.9	16.0	19.7[3]	8.6	7.8	0.3	2.0
Mar 13	37.0	40.4	27.1	28.9	15.0	15.2	0.6	4.8[3]
Apr 13	26.8	30.1[3]	19.0	23.0[3]	11.4	13.8[3]	0.8	4.6[3]
May 10	29.3	31.8	22.6	22.8	14.8	15.8	1.9	7.1[3]
Jun 08	22.4	23.4	17.9	18.8	12.3	12.5	2.3	5.8[3]
Jul 11	31.8	33.7[3]	28.1	27.5	19.0	25.6	8.2	16.7[3]
Aug 10	54.2	58.8[3]	42.8	44.8	33.9	37.4	20.8	29.8[3]
Sep 11	48.2	47.5	35.6	35.8	30.1	35.9[3]	22.1	26.3

[1] Mean of 4 replications with 4 sub-plots.

[2] Significant differences occurred between oil level treatments (pooled inner and outer sub-plots) for each date as determined by analysis of variance F-tests ($P < 0.01$) and Duncan's multiple range test ($P < 0.05$).

[3] Significant differences between inner and outer sub-plots for the oil level as determined by analysis of variance F-tests ($P < 0.05$).

coverage of foliage, with the most severe damage occurring with complete plant coverage. No green stems occurred in plots with entire foliage coverage by oil in September, 1983 and few green stems were counted in these plots through June, 1984. The number of stems in all plots increased greatly from July to August, 1984.

Only in plots with entire foliage coverage was stem density consistently greater in outer than inner sub-plots. This occurred from March through August, 1984. The data indicated that most, if not all, treated plants had died in plots with entire foliage coverage and that recovery was dependent on rhizome growth into plots from plants outside the treated area. Residual toxicity of sediment evidently was not a problem since rhizomes from surrounding border plants established new stems in plots. Inner sub-plots reached numbers similar to outer sub-plots by September, 1984 (Table 1). With sediment and lower stem treatments, lack of consistent differences between inner and outer sub-plots indicated plants were growing from roots and rhizomes of plants that had survived oil treatment.

Sediment hydrocarbon data indicated that oil did not accumulate in the sediment (Table 2). No significant difference was found between levels of hydrocarbon in treated and untreated plots at 10 or 54 days after oil application.

TABLE 2
Mean* oil (mg/100 g dry sediment) remaining in sediment in plots treated with No. 2 fuel oil on September 2, 1983.

	Liters per m^2			
Date	Control 0 l/m^2	Sediment 1.0 l/m^2	Lower Stems 1.5 l/m^2	Entire Stems 2.0 l/m^2
Sep 12**	56 (8.7)***	70 (7.6)	58 (10.6)	53 (15.3)
Oct 26**	59 (13.7)	83 (24.4)	104 (14.7)	128 (24.0)

* Mean of four replications.
** No siqnificant difference ($P < 0.10$) between the control and the treated plots.
*** Standard error of mean.

Some oil accumulated in roots and rhizomes after spraying foliage with No. 2 fuel oil in the second set of experiments. In oil treated plots, a mean of 675 ± 32.9 mg oil/100 g root material occurred versus 337 ± 54.7 mg oil/100 g in untreated plots. The difference was statistically significant at $P < 0.05$.

DISCUSSION

Our data and previous studies (Hampson and Moul 1978; Hershner and Lake 1980; Webb, *et al.* 1985) show that No. 2 fuel oil is toxic to plants. Several

factors indicated that the damage was associated with the amount of plant surface coated with oil. These include: the rapid (1 day) change from green to straw color for completely covered plants, the slight discoloration with partial plant coverage; and the decrease in live stem numbers with increased coverage of plants by oil. No live stems were found in the severely impacted plots since complete death had occurred. Previous studies by Webb, *et al.* (1985) and Alexander and Webb (1985) have shown that plants are most seriously affected when complete coverage of plants occurs, particularly with No. 2 fuel oil.

There are many variables that can alter the effects of toxic chemicals. Hydrocarbon toxicity to plants is generally highest for aromatics and lowest for alkanes (Havis 1950). No. 2 fuel oil typically has an aromatic content of 20-25 percent (National Research Council 1985). The major aromatic components of the No. 2 fuel oil used in this study were alkyl-substituted naphthalenes (Webb, *et al.* 1985).

The principle damage to plants appeared to be due to No. 2 fuel oil contact with foliage. Despite the severe initial damage to foliage material, the failure of plants to regenerate new stems may be partly caused by the translocation of toxic components from foliage to roots and rhizomes. When oil was applied to foliage, the significantly greater amount of hydrocarbons in roots and rhizomes in oil-treated versus control plots seems to indicate that oil was translocated from leaves to roots. The fact that plants were not severely damaged when oil only covered lower plant surfaces seems to indicate contact with leaves could be a necessity for translocation. The presence of hydrocarbons in roots and stems of *S. alterniflora* after low levels of No. 2 fuel oil entered the marsh on flood waters was reported by Hershner and Lake (1980). However, these authors reported that accumulations of hydrocarbons occurred only in dead roots and rhizomes. Destructive effects of contact herbicides are generally restricted to the plant material contacted whereas translocated type herbicides must be transported through the phloem or xylem to other plant areas to cause damage. The real physiological mechanism of death was not determined in this study, but we can conclude that the level of plant coverage is important and that accumulation of hydrocarbons in the root system did occur to some extent and may be an important pathway of damage.

In our study, damage that occurred with only sediment coverage indicated that some No. 2 fuel oil must have penetrated the sediment to reach rhizomes and roots. However, lack of significant differences in hydrocarbons in the sediment between treated plots and control plots 10 days after treatment indicated that oil did not penetrate in significant amounts. Previous studies by Webb, *et al.* (1985) and Alexander and Webb (1985) also indicated little accumulation of fuel oil in the sediment. The greater number of stems in outer sub-plots of 2 L plots compared to inner sub-plots from March through August, 1984 indicated that recovery was occurring by rhizome growth from the adjacent marsh. The regrowth into plots demonstrated that toxic compo-

nents were not present in the sediment in sufficient quantity to prevent regrowth or had been reduced over time to non-toxic levels.

Although sediment toxicity did not appear to be a problem in our study, high concentrations of No. 2 fuel oil can accumulate in sediment and cause death of *S. alterniflora.* Teal *et al.* (1978) reported heavy concentrations of hydrocarbons in the top 6-7 cm of sediment after two No. 2 fuel oil spills in Buzzards Bay, Massachusetts. In areas with heavy concentrations of No. 2 fuel oil in the sediment, *S. alterniflora* regrowth was curtailed (Teal, *et al.* 1978). Booker (1987) suggested that No. 2 fuel oil reduces root viability of *S. alterniflora* by affecting the permeability of root tissue cell membranes. However, there is some indication that damage does not occur by uptake of hydrocarbons through the root system in *S. alterniflora.* Lytle and Lytle (1987) reported that *S. alterniflora* showed no evidence of uptake while *Juncus roemerianus* did take up large amounts of hydrocarbons. Hershner and Lake (1980) also indicated that living *S. alterniflora* does not take up hydrocarbons through the roots.

Sediment type and magnitude of the spill may be important in determination of No. 2 fuel oil effects on salt marshes dominated by *S. alterniflora.* The sediments in marshes of Buzzards Bay, Massachusetts, that were affected by the west Falmouth oil spill in 1969 and the Bouchard 65 barge spill in 1974, apparently were predominantly peat (Hampson and Moul 1978) or mud and peat (Teal, *et al.* 1978). Hampson and Moul (1978) stated that the substrate acted as a sponge and became heavily impregnated with oil due to porosity of the marsh and to interstitial absorption. Sediment in the Galveston Bay marsh, in which this study was performed, is predominantly clay. Little peat occurs because of the warm climate that causes organic matter to decompose rapidly. Oil probably did not penetrate the clay soil readily except where fiddler crab (*Uca* sp.) burrows occur. In only one instance have we noted significant accumulations of oil in the clay soils of Galveston Bay. Large amounts of oil remained on the marsh surface in excess of one week in the absence of tidal dispersion (Alexander and Webb 1987).

The small amount of oil applied in our study may have dispersed more rapidly with flood tides than the spill in Massachusetts. Since No. 2 oil has low viscosity, apparently it is washed easily from the sediment by flooding tidal waters. The studies of Hampson and Moul (1978); Sanders, *et al.* (1977); Teal, *et al.* (1978) and Holt, *et al.* (1978) indicate that a large concentration of oil cannot dissipate rapidly. Teal, *et al.* (1978) reported oil slicks on the water surface of flood tides several years after the contamination of a salt marsh. The oil can penetrate or be buried by sediments as they are reworked by wave action.

The greater number of stems in outer sub-plots than inner sub-plots in plots with entire foliage coverage indicated that recovery occurred primarily by rhizome growth into the plots by undamaged plants outside the plots. A significant realization of the lack of regrowth in inner sub-plots was that with a large spill there would not be plants to recolonize. A good example of the

results of lack of plant recolonization is the erosion of the salt marsh damaged by the Bouchard barge No. 2 fuel oil spill in Massachusetts. Little reestablishment by seedling or rhizome growth of *S. alterniflora* occurred over 3 years (Hampson and Moul 1978). Wave action after removal of stems probably hampered seedling establishment, since plant stems play an important wave dampening role in estuaries (Knutson, *et al.* 1981). However, hydrocarbons in the sediment could have been a significant reason for lack of colonization by seedlings.

The implication, if death of plants occurs because of a large spill, is that there will be no plants to allow recolonization to occur. Although natural reseeding could occur, the surest method to rehabilitate the marsh would be transplanting. Immediate replanting would allow the standing dead stems to provide some wave protection as transplants became established. Transplanting techniques for *S. alterniflora* have been documented for the Atlantic and Gulf coasts of the U.S. (Lewis 1982). The limiting factor for plants would be hydrocarbon accumulation in the sediment. Sediment analyses could be quickly accomplished to determine if hydrocarbons were present in quantities to cause problems to plants. Brown, Pabst and Light (1978) have reported a rapid thin-layer chromatographic technique for estimating the relative concentration of crude oil in sediments. Total hydrocarbons also can be estimated by the methods used in this study.

CONCLUSIONS

Severity of damage to *S. alterniflora* by No. 2 fuel oil is directly related to the amount of plant coverage. Direct physical damage to contacted tissues plus translocation of toxic components of the oil from stems to the root system causes death or a reduction in the ability of the root system to regenerate shoots. Soil penetration also can cause some damage to roots, which may result in biomass and stem reductions, but oil contact with foliage is the major cause of death and long-term damage in Gulf Coast marshes. Because of the rapid and severe damage of No. 2 fuel oil, every effort should be made by oil spill control personnel to prevent the oil from entering salt marshes. Large quantities that do enter and cause damage should be removed as quickly and thoroughly as possible to prevent penetration of and accumulation in sediment and plant tissues. The long-term effects on regrowth can be evaluated within days or weeks because of the obvious immediate effects of this oil (absence of green color and lack of shoot regrowth). When death of most plant growth occurs and regeneration does not occur, transplantation of the marsh should be accomplished if significant quantities of oil are not present in the sediment. Although recolonization through seedling establishment is possible, natural colonization may not occur because of high wave energy in many areas.

ACKNOWLEDGMENTS

Funding for this study was provided by the State of Texas through the Center for Energy and Mineral Resources at Texas A&M University and by the Harris and Eliza Kempner Fund. Oil used in this study was donated by Exxon USA.

LITERATURE CITED

ALEXANDER, S.K. and J.W. WEBB. 1985. Seasonal response of *Spartina alterniflora* to oil. p. 355-357. In *Proceedings 1985 Oil Spill Conference.* American Petroleum Institute, Washington, DC.

_____ and _____. 1987. Relationship of *Spartina alterniflora* growth to sediment oil content following an oil spill. p. 445-449. In *Proceedings 1987 Oil Spill Conference.* American Petroleum Institute, Washington, DC.

BAKER, J.M. 1970. The effects of oil on plants. *Environmental Pollution.* 1:27-44.

BECCASIO, A.D., N. FOTHERINGHAM, A.E. REDFIELD, R.L. FREW, W.M. LEVITAN, J.E. SMITH and J.O. WOODROW, Jr. 1982. Gulf coast ecological inventory guide and information base. U.S. Fish and Wildlife Service. **FWS/OBS-82-55.** 191 pp.

BOOKER, F. 1987. Fuel oil effects on membrane permeability in *Spartina alterniflora* roots. p. 469-471. In *Proceedings 1987 Oil Spill Conference.* American Petroleum Institute, Washington, DC.

BROWN, L.R., G.S. PABST and M. LIGHT. 1978. A rapid field method for detecting oil in sediments. *Marine Pollution Bulletin.* **9(3)**:81-83.

BURNS, K.A. and J.M. TEAL. 1971. Hydrocarbon incorporation into the salt marsh ecosystem from the West Falmouth oil spill. Technical Report 71-69. Woods Hole Oceanographic Institution.

CONNER, J.V. and F.M. TRUESDALE. 1972. Ecological implication of a fresh water impoundment in a low salinity marsh. p. 259-276. In R.H. Chabreck (ed.) *Proceedings Coastal Marsh Estuary Management Symposium.* Louisiana State University, Baton Rouge, Louisiana.

CROW, S.A., Jr. 1974. Microbial aspects of oil intrusion in the estuarine environment. Ph.D. Thesis, Louisiana State University. Baton Rouge, Louisiana. 179 pp.

DAVIDSON, R.B. and R.H. CHABRECK. 1983. Fish, wildlife, and recreational values of brackish marsh impoundments. p. 89-114. In R.J. Varnell (ed.) *Proceedings Water Quality and Wetland Management Conference.* New Orleans, Louisiana.

DELAUNE, R.D., W.H. PATRICK, Jr. and R.J. BURESH. 1979. Effects of crude oil on a Louisiana *Spartina alterniflora* salt marsh. *Environmental Pollution.* **20**:21-33.

HAMPSON, G.R. and E.T. MOUL. 1978. No. 2 fuel oil spill in Bourne, Massachusetts: immediate assessment of the effects on marine invertebrates and a 3 year study of growth and recovery of a salt marsh. *Journal of the Fisheries Research Board of Canada.* **35**:731-734.

HAVIS, J.R. 1950. Herbicidal properties of petroleum hydrocarbons. Agricultural Experimental Station Memoirs, No. 298. Cornell University, Ithaca, New York. 20 pp.

HERKE, W.H. 1971. Use of natural and semi-impounded, Louisiana tidal marshes as nurseries for fishes and crustaceans. Ph.D. Thesis, Louisiana State University, Baton Rouge, Louisiana. 242 pp.

HERSHNER, C. and J. LAKE. 1980. Effects of chronic pollution on a salt marsh grass community. *Marine Biology.* **56**:163-173.

_____ and K. MOORE. 1977. Effects of the Chesapeake Bay oil spill on salt marshes of the lower bay. p. 529-533. In *Proceedings 1977 Oil Spill Conference.* American Petroleum Institute, Washington, DC.

HOLT, S., S. RABALAIS, N. RABALAIS, S. CORNELIUS and J.S. HOLLAND. 1978. Effects of an oil spill on salt marshes at Harbor Island, Texas. I. Biology, p. 345-352. In *Conference: Assessment of the Ecological Impacts of Oil Spills.* American Institute of Biological Scientists.

KNUTSON, P.L., J.C. FORD and M.R. INSKEEP. 1981. National survey of planted salt marshes. *Wetlands*. **1**:129-157.

KREBS, C.T. and C.E. TANNER. 1981. Cost analysis of marsh restoration through sediment stripping and *Spartina* propagation. p. 375-385. In *Proceedings 1981 Oil Spill Conference*. American Petroleum Institute, Washington, DC.

LEWIS. R.R., III. 1982. *Creation and Restoration of Coastal Plant Communities*. CRC Press, Inc. Boca Raton, Florida.

LINDALL, W.N., Jr. and C.H. SALOMAN. 1977. Alteration and destruction of estuaries affecting fishery resources of the Gulf of Mexico. *Marine Fisheries Review*. **39(9)**:1-7.

LYTLE, J.S. and T.F. LYTLE. 1987. The role of *Juncus roemerianus* in cleanup of oil-polluted sediments. p. 495-501. In *Proceedings 1987 Oil Spill Conference*. American Petroleum Institute, Washington, DC.

McHUGH. J.L. 1968. Are estuaries necessary? *Commercial Fisheries Review*. **30(11)**:37-44.

MOCK, C.R. 1966. Natural and altered estuarine habitats of penaeid shrimp. *Proceedings of the Gulf and Caribbean Fishery Institute*. **19**:86-98.

MONTAGUE, C.L., S.M. BUNKER, E.B. HAINES, M.L. PACE and R.L. WETZEL. 1981. Aquatic macroconsumers. p. 69-85. In L.R. Pomeroy and R.G. Weigert (eds.), *The Ecology of a Salt Marsh*. Springer-Verlag, New York.

NATIONAL RESEARCH COUNCIL. 1985. *Oil in the Sea: Inputs, Fates and Effects*. National Academy of Sciences, Washington, DC.

ODUM, W.E., J.C. ZIEMAN and E.J. HEALD. 1972. The importance of vascular plant detritus to estuaries, p. 93-112. In R.H. Chabreck (ed.) *Proceedings of the Coastal Marsh and Estuary Management Symposium*. Louisiana State University, Baton Rouge, Louisiana.

PETERS, D.S., D.W. AHRENHOLZ and J.T. RICE. 1978. Harvest and value of wetland associated fish and shellfish. In P.E. Greeson, J.R. Clark and J.E. Clark (eds.) *Wetland Functions and Values: the State of our Understanding*. American Water Resources Association. Urbana, Illinois. 606 pp.

SANDERS, H.L., J.F. GRASSLE, G.R. HAMPSON, L. MORSE and S. PRICE. 1977. Anatomy of an oil spill: the west Falmouth study. Report to U.S. Environmental Protection Agency. 574 pp.

SAS INSTITUTE, INC. 1985. SAS User's Guide: Statistics, Version 5 Ed. SAS Institute Inc., Cary, North Carolina. 956 pp.

SCIFRES, C.J. 1980. *Brush Management, Principles and Practices for Texas and the Southwest*. Texas A&M University Press. College Station. Texas. 360 pp.

TEAL, J.M., K. BURNS and J. FARRINGTON. 1978. Analyses of aromatic hydrocarbons in intertidal sediments resulting from two spills of No. 2 fuel oil in Buzzards Bay, Massachusetts. *Journal of the Fisheries Research Board of Canada*. **35**:510-520.

WEBB, J.W., S.K. ALEXANDER and J.K. WINTERS. 1985. Effects of autumn application of oil on *Spartina alterniflora* in a Texas salt marsh. *Environmental Pollution*, Series A. **38(4)**:321-337.

_____, G.T. TANNER and B.H. KOERTH. 1981. Oil spill effects on smooth cordgrass in Galveston Bay, Texas. *Contributions in Marine Science*. **24**:107-114.

ZIMMERMAN, R.J. and T.J. MINELLO. 1984a. Fishery habitat requirements: utilization of nursery habitats by juvenile penaeid shrimp in a Gulf of Mexico salt marsh. p. 371-383. In B. J. Copeland, K. Hart, N. Davis and S. Friday. (eds.) *Proceeding on Research for Managing the Nation's Estuaries*. National Sea Grant College Program.

_____ and _____. 1984b. Densities of *Penaeus aztecus*, Penaeus setiferus, and other natant macrofauna in a Texas salt marsh. *Estuaries*. **7(4a)**:421-433.

_____, _____ and G. ZAMORA, Jr. 1984. Selection of vegetated habitat by brown shrimp, *Penaeus aztecus* in a Galveston Bay salt marsh. *Fishery Bulletin*. **82(2)**:325-336.

Contributions in Marine Science. (1991) Vol. 32.

A NEW RECORD OF *PAULINELLA OVALIS*: FILOSEA: EUGLYPHINA

Susan E. Safford[1]

Gulf Coast Research Laboratory
Ocean Springs, Mississippi 78546

ABSTRACT

Monthly nanoplankton samples (2-20μm) were collected over a 13 month period during 1977 and 1978 in St. Louis Bay, Mississippi to determine the seasonal algal composition of the bay. The testate rhizopod, *Paulinella ovalis*, originally identified by Wulff (1919) as the chrysophycean alga, *Calycomonas ovalis*, was a major component of samples from July - November 1978, with volumes ranging from 18-78% of the total. This is the first record of its occurrence in the Gulf of Mexico and adjacent waters.

INTRODUCTION

The testate rhizopod, *Paulinella ovalis* n. comb. Johnson, Hargraves, and Sieburth (1988), (formerly *Calycomonas ovalis* Wulff 1919) was a common form in nanoplankton samples collected from St. Louis Bay, Mississippi during 1977 and 1978. Its abundance ranged from < 1% of the total volume to as high as 78% of the volume. This is the first record of *P. ovalis* from the Gulf of Mexico. This species, formerly *C. ovalis*, was first described by Wulff (1919) in the Bay of Kiel in the North Sea, and in the Barents Sea. It has most frequently been reported from the North Atlantic (Hulbert 1965; Marshall and Wheeler 1965; Throndsen 1969; Carpenter 1971; Campbell 1973; Parke and Dixon 1976; Stanley and Daniel 1985; Sieburth, Johnson, and Hargraves 1988), with one record from New Zealand (Throndsen 1969). The organism found in St. Louis Bay was oval shaped, 4-6μm in length, had 4-5 annular rings and no chromatophore nor observed flagellum, and was surrounded by a lorica with one opening at the narrow end (Fig. 1). This fits the description of *P. ovalis* found in Johnson, *et al.* (1988). An additional feature of *P. ovalis*, which was noted in some live specimens from our samples, is the presence of a pseudopodium extended through the narrow opening of the lorica.

MATERIAL AND METHODS

Water samples were collected in Niskin bottles (General Oceanics, Inc., Miami, Florida) at selected sites and standard depths (0.25m, 0.8m, 1.4m, and 2.0m) (Fig. 2). St. Louis Bay is very

[1] Present address: The University of Texas at Austin, Marine Science Institute, POB 1267, Port Aransas, Texas 78373

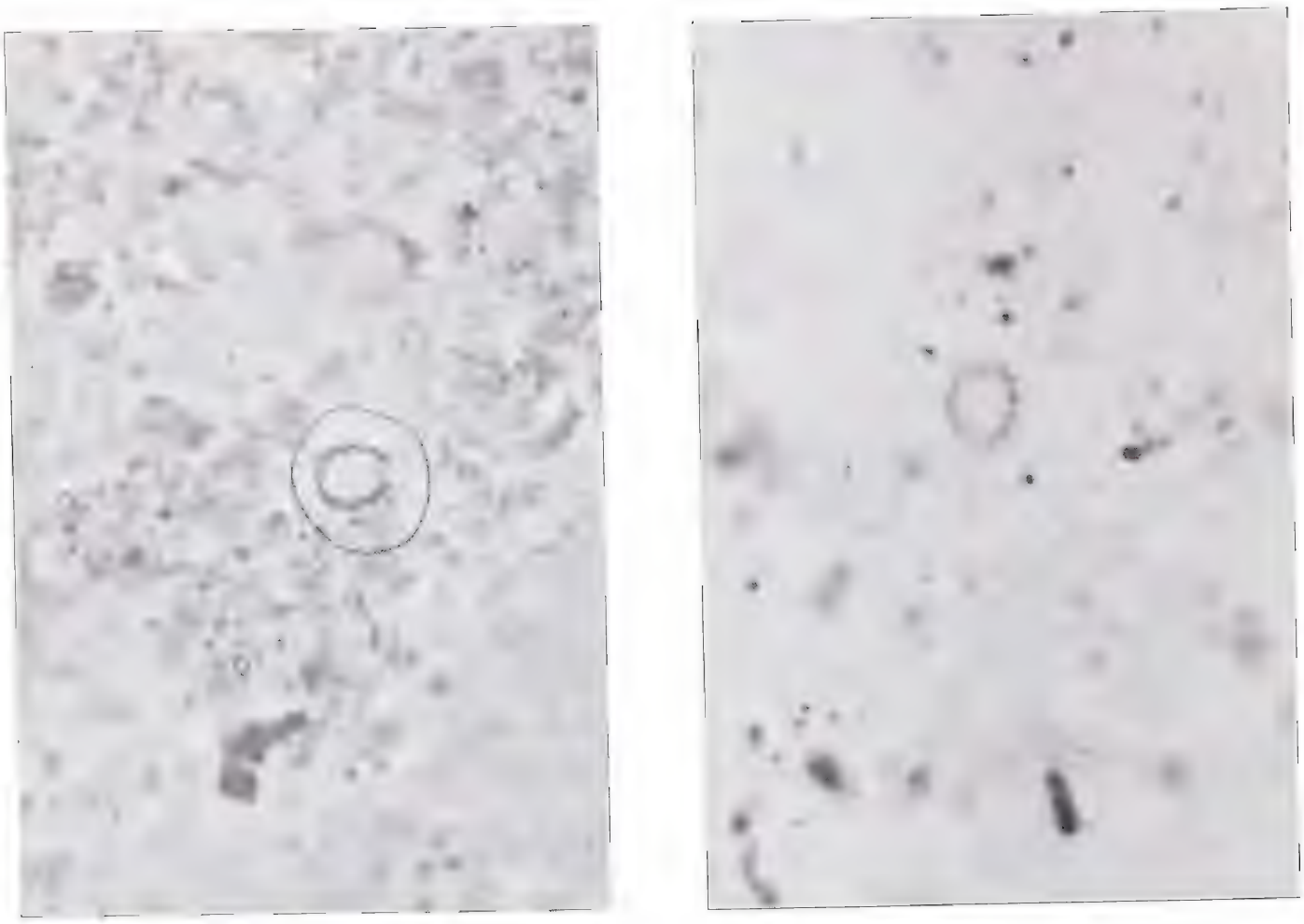


FIG. 1. A lateral view of *Paulinella ovalis* showing the characteristic lorica and annular rings. Positive phase contrast. 1000× magnification.

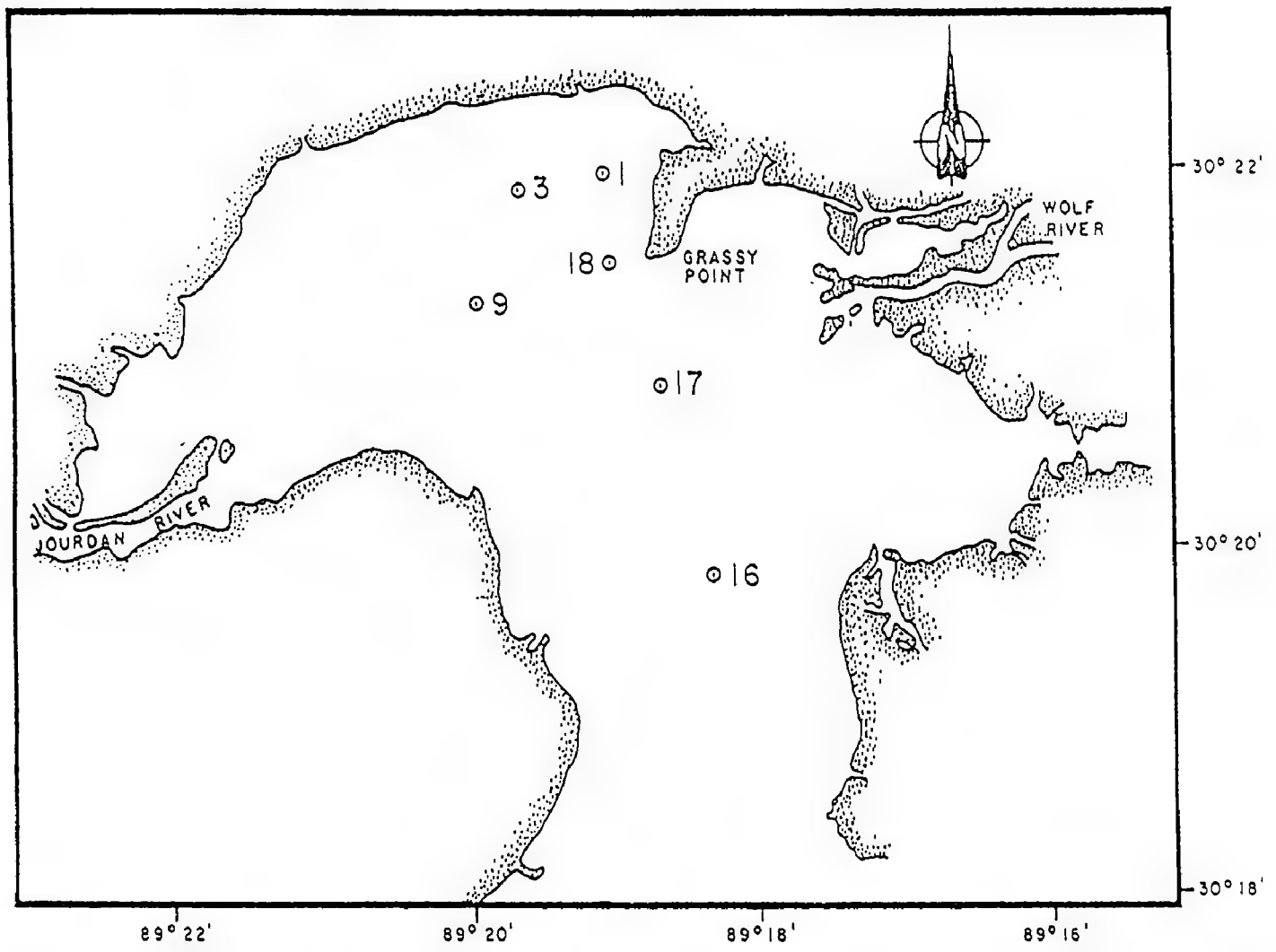


FIG. 2. St. Louis Bay station locations from which nannoplakton samples were collected.

shallow, so only 2 depths, 0.25 and 0.8m, were sampled at stations 1, 3, and 9. The 1.4m sample was taken most months from stations 16, 17, and 18, and occasionally a 2.0m sample was obtained from station 16. A nanoplankton sample was obtained by filtering 5 liters of water through a 20μm mesh net, then filtering an aliquot (125-400 ml) of the 20μm water through a 0.8μm Millipore filter (Millipore Corp., Bedford, Massachusetts). The filters were fixed in cedar balsam, cut to size, and affixed to microscope slides for analysis. A Zeiss positive phase contrast microscope with a 1000× oil immersion lens (Zeiss, Carl, Inc., Thornwood, New York) was used for all nanoplankton identifications. Temperature and salinity were taken with a Beckman portable salinometer (Beckman Instruments, Inc., Palo Alto, California).

RESULTS

Average salinity and water temperature plotted against the number of tests/L for each sampling month are found in figures 3a and 3b, respectively. The population of *P. ovalis* was low during the winter months, but specimens were present at some stations every month. *P. ovalis* was most abundant from July to November 1978, when it increased from < 1% of the total volume to 43% in November, and comprised as much as 78% of the volume at some individual depths. During these months, salinity averaged over all stations and depths ranged from 8.5 - 16.5 ‰, with salinities from individual depths as low as 5.6 ‰

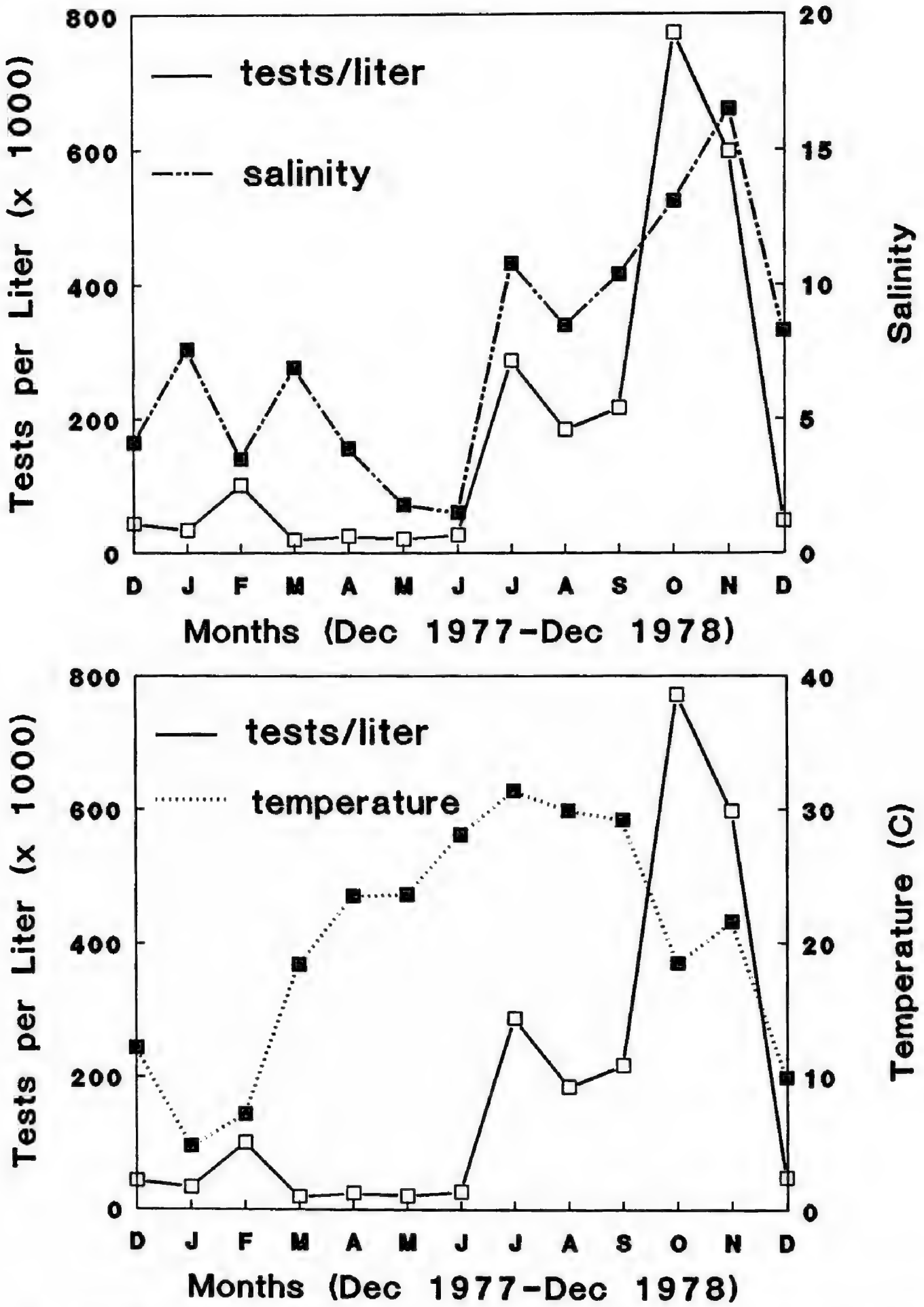


FIG. 3. A strong correlation is seen between salinity levels and number of tests of *Paulinella ovalis* (FIG. 3A), while no strong relationship is seen between temperature and biomass (FIG. 3B) during the sample period.

and as high as 17.1 ‰. The mean temperature during these months, averaged over all collections, ranged from 18.5 - 31.4° C, while temperatures from individual depths ranged from 18.2 - 31.6° C.

DISCUSSION

Little ecological information is available on *P. ovalis*, though all the localities from which it has been reported are in or near estuaries, which are usually sites of high productivity. There are only a few records of its occurrence and even fewer reports on the environmental conditions associated with it. Until the present study, the most southerly reported occurrence in the United States of *P. ovalis* has been off the North Carolina coast (Carpenter 1971; Campbell 1973; Stanley and Daniel 1985). The only other record of its occurrence outside the North Atlantic is the report of its presence in Wellington Harbor, New Zealand (Throndsen 1969). Apparently *P. ovalis* is tolerant of extreme salinities and temperatures as Carpenter (1971) found *P. ovalis* in waters with salinities of 1.7-35 ‰ and temperatures ranging from 4.5 - 30° C. This temperature range is similar to that reported here, 4.7 - 31.7° C. Also, the salinities recorded by Carpenter (1971) are similar to those reported by Campbell (1973), 0-31 ‰, and those found in this study, 0.3-18.8 ‰. However, these cited studies give little information on the relative abundance of *P. ovalis* at these different temperatures and salinities.

In the present study, large numbers of *P. ovalis* were found immediately after a large increase in salinity (9 ‰ increase in 1 month) and in association with warm (> 20° C) water temperatures. Conversely, the population decreased 10-fold over a one month period when salinity dropped precipitously. Thus, salinity seems to be an important determinant in predicting population size as populations remained small when water temperatures were high (> 20° C) and salinities were low (< 10 ‰). Presumably, the relationship between salinity and population is related to the influx of Gulf water on its food organisms, probably cyanobacteria. Sieburth *et al.* (1988) found that *P. ovalis* reached a population size of 6×10^6 tests/L when it was feeding on cyanobacteria associated with a summer bloom of *Aureococcus anophagefferens*. The limited information available suggests that good habitats for *P. ovalis* are coastal bays with freshwater inflow during months when water salinity, temperature and productivity are high.

ACKNOWLEDGMENTS

I would like to thank the following people: Dr. Peter Campbell, University of North Carolina at Chapel Hill and Mr. William Gardiner, University of South Florida who confirmed the identity of this organism; Ms. Lois Coquet and Mr. Walter Brehm, Gulf Coast Research Laboratory who provided me with the map and the data on *Paulinella ovalis*; Dr. Robert Woodmansee, who was in charge of this project, for advice and encouragement; and E.I. du Pont de Nemours and Company

whose contract made this study possible. The University of Texas at Austin Marine Science Institute Contribution No. 821.

LITERATURE CITED

CAMPBELL, P.H. 1973. Studies on brackish water phytoplankton. pp 200, 310-316. University of North Carolina Sea Grant Publication UNC-SG-73-07.

CARPENTER, E.J. 1971. Annual phytoplankton cycle of the Cape Fear River Estuary, North Carolina. *Chesapeake Science.* **12(2)**:95-104.

HULBURT, E.M. 1965. Flagellates from brackish waters in the vicinity of Woods Hole, Massachusetts. *Journal of Phycology.* **1**:87-94.

JOHNSON, P.W., P.E. HARGRAVES, and J. McN. SIEBURTH. 1988. Ultrastructure and ecology of *Calycomonas ovalis* Wulff, 1919, (Chrysophyceae) and its redescription as a testate rhizopod, *Paulinella ovalis* N. Comb. (Filosea:Euglyphina). *Journal of Protozoology.* **35(4)**:618-626.

MARSHALL, N. AND B.N. WHEELER. 1965. Role of the coastal and upper estuarine waters contributing phytoplankton to the shoals of the Niantic estuary. *Ecology.* **46(5)**:665-673.

PARKE, M. AND P.S. DIXON. 1976. Checklist of British marine algae. Third Revision. *Journal of the Marine Biological Association of the United Kingdom.* **56**:527-594.

SIEBURTH, J. McN., P.W. JOHNSON, AND P.E. HARGRAVES. 1988. Ultrastructure and ecology of *Aureococcus anophagefferens* gen. et sp. nov. (Chrysophyceae): the dominant picoplankter during a bloom in Narragansett Bay, Rhode Island, summer 1985. *Journal of Phycology.* **24**:416-425.

STANLEY, D.W. AND D.A. DANIEL. 1985. Seasonal phytoplankton density and biomass changes in South Creek, North Carolina. *Journal of the Elisha Mitchell Society.* **101(2)**:130-141.

THRONDSEN, J. 1969. Flagellates of Norwegian coastal waters. *Nytt Magasin for Botanik.* **16**:161-216.

WULFF, A. 1919. Uber das kleinplankton der Barentssee. *Wissenschaftliche Meeresuntersuchungen der Kommission zur Wissenschaftlichen Untersuchung der Deutschen Meere, Abteilung Helgoland.* **13**:97-113.

Contributions in Marine Science. (1991) Vol. 32.

FACTORS INFLUENCING THE STANDING CROP OF DIATOM EPIPHYTES OF THE SEAGRASS *HALODULE WRIGHTII* ASCHERS. IN SOUTH TEXAS SEAGRASS BEDS

Jerilyn Jewett-Smith

University of North Carolina at Wilmington
Center for Marine Science Research
7205 Wrightsville Ave.
Wilmington, North Carolina 28403

ABSTRACT

The diatom epiphytes on leaf blades of *Halodule wrightii* Aschers. were examined in four South Texas seagrass beds from June 1986 through June 1987. The standing crop of epiphytic diatoms on the blade surfaces of *H. wrightii* was lowest at the hypersaline site in the Upper Laguna Madre (annual means: 345.1 cells mm^{-2}; 39.5 ‰ salinity) and the highest at the mouth of Nueces Bay (annual mean: 3151.8 cells mm^{-2}; 31.4 ‰ salinity). Intermediate standing crop values were recorded at lower Corpus Christi Bay (annual mean: 633.5 cells mm^{-2}; 35.1 ‰ salinity) and Redfish Bay (annual mean: 1451.5 cells mm^{-2}; 30.9 ‰ salinity). The four sites were found to vary significantly (via ANOVA, $p < 0.05$) in the standing crop of epiphytic diatoms on *H. wrightii*. Combining all four sites and using a stepwise multivariate regression, the three interrelated variables of available light (% of surface light at the bottom), water clarity, and depth plus the variable of temperature accounted for 52.95% of the variation seen in the epiphytic diatom standing crop on *H. wrightii*.

INTRODUCTION

Diatoms, which are a major colonizer of newly emerged aquatic macrophyte surfaces (Round 1981), can be studied from a taxonomic or ecological standpoint. Most investigations of marine diatom assemblages in the northern and western Gulf of Mexico have looked at the diatom components from a taxonomic viewpoint in: planktonic communities (Freese 1952, Morgan 1975); benthic communities (Wood 1963, Morgan 1975); epiphytic diatoms on attached benthic algae (Medlin 1983) and on seagrasses from Mississippi Sound (Sullivan 1979). The present study was ecologically-oriented and has measured the diatom standing crops on the marine macrophyte *Halodule wrightii* Aschers. along with concurrent measurement of environmental variables.

Physical and chemical properties of the water overlying marine and freshwater macrophyte beds have been evaluated for possible clues to the environmental determinants of the composition of epiphytic associations. In

Australian marine epiphytic communities, increasing salinity slowed the rate of growth and development of the host seagrass and red algal epiphytes, and decreased percent cover by epiphytes at some sites (Harlin, Woelkerling and Walker 1985). Species richness of the Australian marine epiphytic communities in Harlin's study correlated negatively with salinity across a salinity range of 36 ‰ to 70 ‰. Lowered production at salinities above 35 ‰ is not always the case: Walker (1985) found maximal production of *Amphibolis antarctica* (Labill.) Sonder & Aschers. from Western Australia, at a salinity of 42 ‰.

In the northern Gulf of Mexico, Macauley, Clark and Price (1988) found no significant correlation between epiphyte standing crop on *Thalassia testudinum* Banks ex Konig and light energy or salinity. He did, however, demonstrate a negative correlation between standing crop and water temperature. The range in salinity for the northern Gulf of Mexico was considerably less than the current report or that of Australia's Shark Bay. Host specificity of epiphytes depended upon the fertility of the water body in freshwater lakes of the northern United States and the eastern United Kingdom: in low nutrient lakes the epiphytes were host specific but in high fertility lakes external environmental factors became more important in determining the epiphytic community composition (Eminson and Moss 1980). In the above studies, salinity, light, phosphate concentration and temperature were suggested as factors influencing the composition and abundance of the epiphytic community.

In the present study, several environmental variables were examined along with the diatom epiphytes on the seagrass, *Halodule wrightii* Aschers., from four South Texas seagrass communities. Concurrently with epiphyte standing crop and host shoot density, the following environmental variables were measured: salinity, temperature, depth, light, phosphate, inorganic nitrogen (nitrate + nitrite). A stepwise multivariate regression evaluated the relationship of the measured environmental variables to epiphyte standing crop.

MATERIALS AND METHODS

Four seagrass beds were chosen for study: Redfish Bay (RFB); the mouth of Nueces Bay (MNB); lower Corpus Christi Bay at Fish Pass (LCC); and the Upper Laguna Madre (ULM), (Fig. 1). RFB is located on the east side of Stedman Island at the Fin & Feather fish camp adjacent to the Aransas Channel and is the closest to oceanic input from the Gulf of Mexico. MNB is located on Upper Corpus Christi Bay on the southwest side of Indian Point Park and receives the most influence from freshwater inflow from the Nueces River and the prevailing southeasterly wind (Tony Amos, personal communication, 1989). LCC is located off the west side of Mustang Island in the Mustang State Park at the Fish Pass. ULM is located in the Padre Island National Seashore at Bird Island Basin, on the west side of North Padre Island. These four sites represent a wide range of estuarine seagrass sites in the South Texas area.

At each of the four sites, sample cores were taken monthly from June through September 1986. Three cores (diameter = 8.9 cm, depth = 30 cm) were taken in each seagrass bed from June through September for estimation of shoot density. These cores were sieved in the field, and the plant material returned to the laboratory for further cleaning.

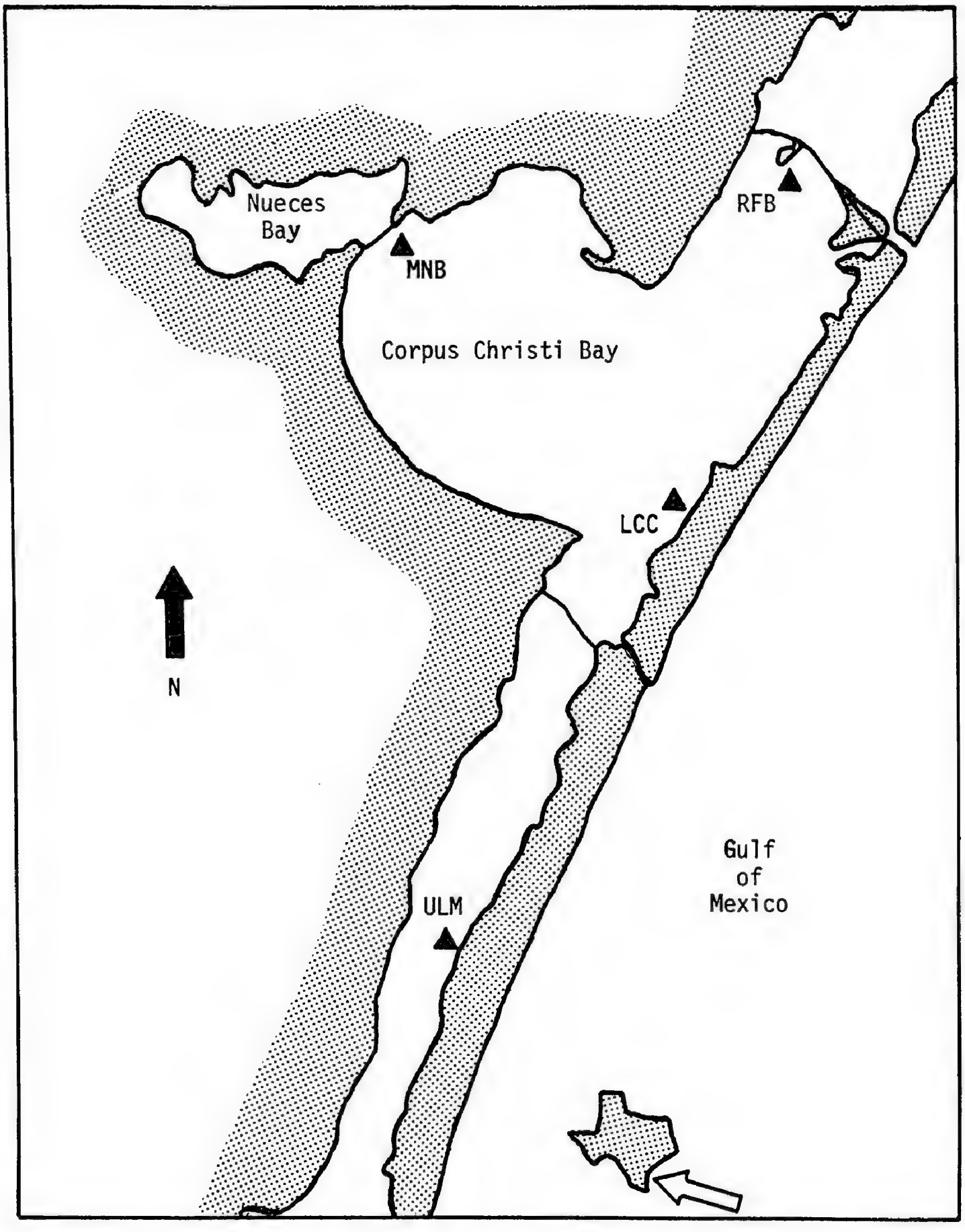


FIG. 1. Map of the study area in South Texas. RFB = Redfish Bay at Fin & Feather Marina on Stedman Island. MNB = Mouth of Nueces Bay at Indian Point Park on Upper Corpus Christi Bay. LCC = Lower Corpus Christi Bay at Fish Pass, Mustang Island State Park. ULM = Upper Laguna Madre at Bird Island Basin, Padre Island National Seashore.

Water samples were taken at each site monthly from June through September 1986 and continued bimonthly from October 1986 through June, 1987. All field sampling took place between 1000 and 1400 hrs. Six of the 10 field trips were completed on the same day. Two trips were completed within 24 hours, two others within 6 days. The samples were filtered immediately in the field (GFA, Millipore, 0.45 μM pore size), transported on ice and frozen for later analysis. Analysis for phosphate involved reacting an aliquot of the filtered seawater with a composite reagent containing 5 mM molybdic acid, 61 mM ascorbic acid and 0.4 mM trivalent antimony, and reading the resultant blue solution at 885 nm in a Gilford model 250 spectrophotometer (Parsons, Maita and Lalli 1984). Inorganic nitrogen analysis as nitrate + nitrite, involved passing an aliquot of seawater through a nitrate reduction column of cadmium filings coated with metallic copper which reduces nitrate to nitrite. The resulting nitrite was diazotized with 58 mM sulfanilamide, coupled with 4.5 mM N-(1-naphthyl)-ethylenediamine to form a pink colored azo dye, which was quantified at 543 nm in a Gilford model 250 spectrophotometer (Parsons, *et al.* 1984).

At each site on each collection date, light measurements in the water column just below the surface of the water, at 30 cm of depth and among the plants at 12 cm off the bottom (approximately mid-blade) were recorded using a LI-COR model LI-185A quantum meter fitted with a submersible flat cosine corrected sensor (model LI-192SA). The light values at each site were converted to two numbers:

(1) percentage of surface light at the bottom as a measure of light available to the plants and

(2) the percentage of surface light at 30 cm of depth as a measure of clarity of the water column.

Along with the sediment cores, water chemistry, and light measurements, the diatom epiphytes of *Halodule wrightii* were collected at each of the four sites. Whole *H. wrightii* shoots were collected in the surrounding bed and preserved in the field in fixative (2.5% glutaraldehyde, 0.1 M cacodylic acid, 0.25 M sucrose). From three preserved shoots, the oldest leaf from each shoot was measured for length and width, and individually dissolved in 3 ml hot concentrated sulfuric acid in a conical centrifuge tube and allowed to stand for 24 hours. The sulfuric acid digest was diluted with distilled water, allowed to settle, and the top solution removed with a pipette. The dilution was repeated until the solution was ≥ pH 7. The cleaned diatoms were concentrated to a constant volume and counted in a Palmer-Maloney plankton counting cell. Counts were normalized to give a density (or standing crop) of diatoms per mm^2 of blade surface using the length and width information for each blade.

A stepwise multivariate regression (SAS 1987) was tested on all variables (except shoot density) with diatom epiphyte standing crop as the dependent variable. Separate multiple regression analyses were run including the shoot density because shoot density measurements were only taken between June and September 1986. Two-way ANOVA procedures were used to determine if there was a significant difference between sites and dates on the shoot density of *H. wrightii* or on diatom epiphyte standing crop.

RESULTS

Physico-chemical Variables

The variables of temperature and salinity were good descriptors of seasonal fluctuations in the water column. Temperatures in the winter during mid-day ranged as low as 17.0° C and reached summer highs of 33.0° C, with an annual mean of 26.5° C. Salinity patterns tended to track the temperature pattern: high salinities during summer months, low salinities during winter months (Fig. 2). The most saline site throughout the investigation was ULM (maximum salinity during September 1986 = 46.0 ‰; minimum salinity during May, 1987 = 30.0 ‰; mean = 39.5 ‰. LCC (annual salinity mean = 35.1 ‰) was typically intermediate between the Laguna Madre site and the other two

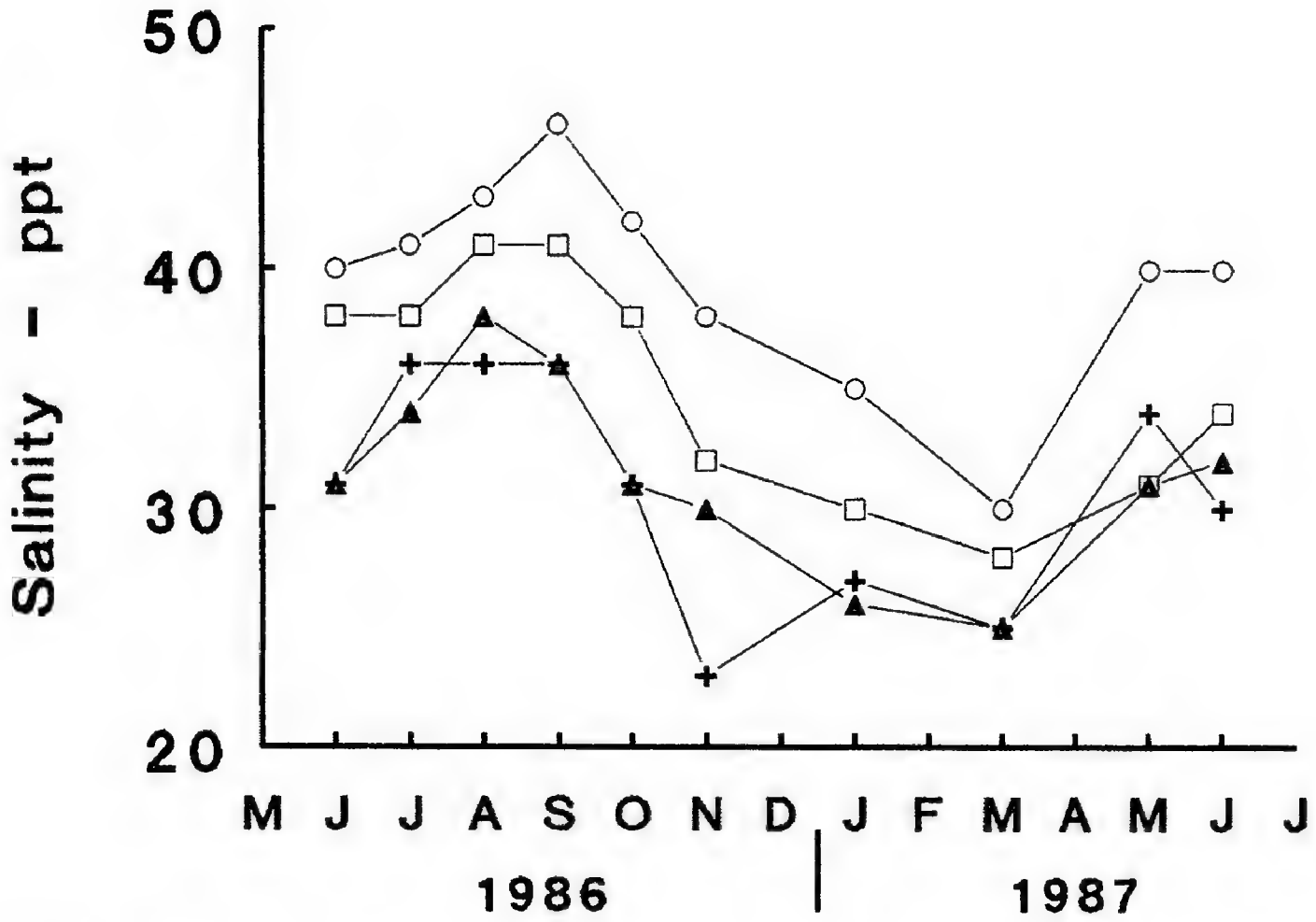


FIG. 2. Refractometer measurement of salinity at the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). There was a significant difference in salinity between sites (ANOVA, $p > F = .0003$), with the mean of the Upper Laguna Madre site significantly higher than the mouth of Nueces Bay and Redfish Bay sites, but not significantly higher than the Lower Corpus Christi site tested via Sheffe's multiple comparison procedure.

sites: MNB, annual mean = 31.4 ‰; RFB, annual mean = 30.9 ‰. The deepest *H. wrightii* beds were at ULM (Fig. 3) with a mean depth of 92.0 cm of water. The other three sites had very similar depths: MNB with an annual mean of 31.0 cm; LCC with an annual mean of 30.2 cm; RFB with an annual mean of 33.5 cm. Of the four sites only MNB was exposed to air during an exceptionally low tide in January 1987.

Water clarity, measured as % surface light at 30 cm depth, did not exhibit a seasonal pattern (Fig. 4). MNB was the lowest with an annual mean of 49.6% and exhibited the most variability in water clarity (low = 16.7%, high = 71.3%). ULM had an annual mean water clarity of 72.5%, the highest of the four sites.

Unlike clarity, available light, as % surface light at the bottom was quite similar between the four sites (Fig. 5). The MNB site was the most variable in available light (low = 16.7%, high = 100% during exposure to air at a low tide). Upper Laguna Madre, the deepest site, varied between a low available light of 34.1% and a highof 58.1% with a mean of 51.6% at the bottom. Most of the measurements in the four seagrass beds gave 50.0% or more of the

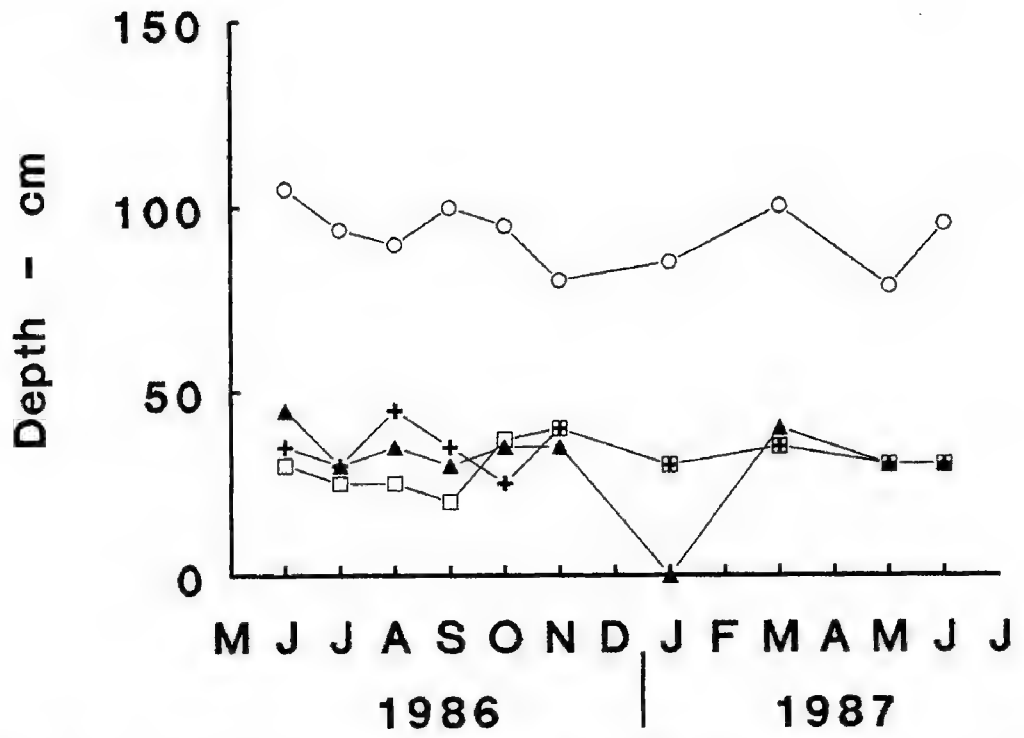


FIG. 3. Measurement of depth of the *Halodule wrightii* seagrass beds at the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). There was a significant difference in depth between sites (ANOVA, $p > F = .0001$), with the mean at the Upper Laguna Site significantly higher than the means at the other three sites via Scheffe's multiple comparison procedure.

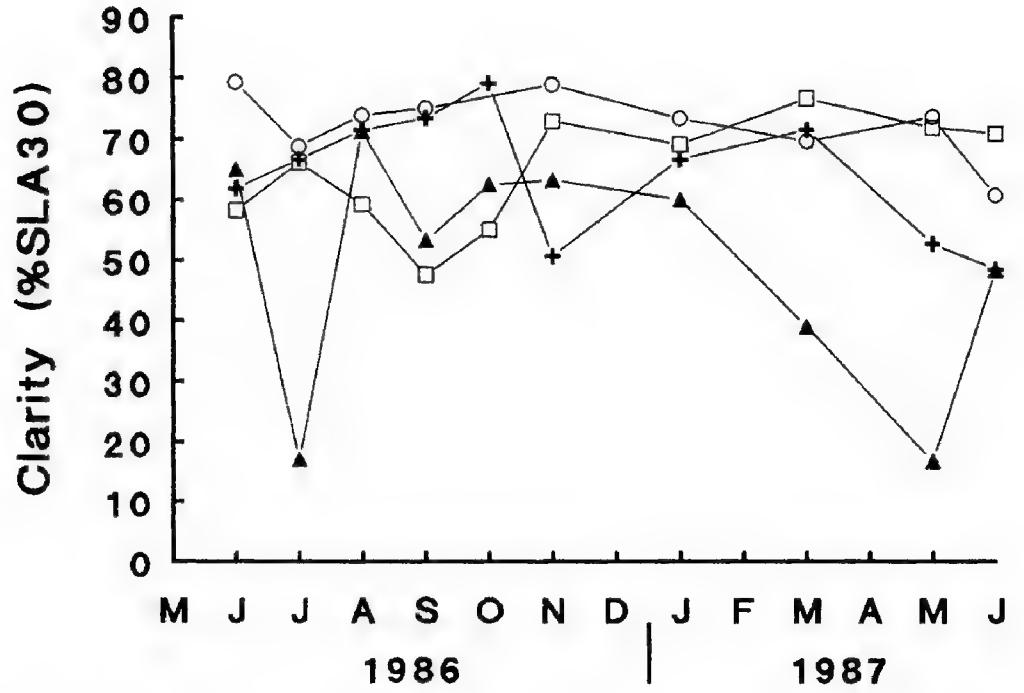


FIG. 4. Water clarity as measured by % of the surface light at 30 cm. of depth (%SLA30) over seagrass beds of *Halodule wrightii* at the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). There was a significant difference in water clarity between the four sites (ANOVA $p > F = .0029$) with the annual mean of the Upper Laguna Madre significantly higher than that at the mouth of Nueces Bay site as tested by Scheffe's multiple comparison procedure.

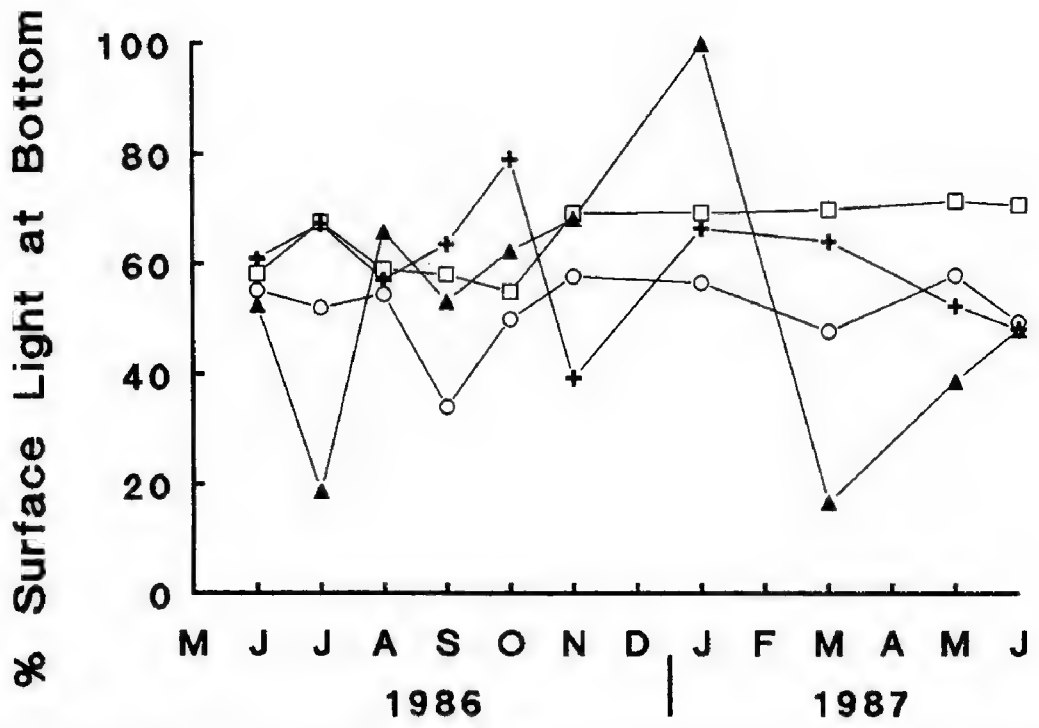


FIG. 5. Available light as measured by the % of light at the bottom (measured at 12 cm from the base of the *Halodule wrightii* shoots) within the seagrass bed at the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). There was not a significant difference in available light between the four sites (ANOVA $p > F = .1365$).

surface light available at the bottom at the time of sampling (usually from 10:00 AM to 2:00 PM). The average mid-day photon flux fluence rate measured during this study was between 1500 and 2000 μmol $m^{-2} \cdot s^{-1}$.

Nutrients values were not good descriptors of seasonality in the seagrass bed. The lowest mean PO_4 values were from RFB (annual mean = .438 μM, minimum = .208 μM, maximum = 0.744 μM, Fig. 6) and the highest at LCC (annual mean = .834 μM, maximum = 2.098 μM, minimum = .225 μM).

The other inorganic nutrient in this study, nitrogen (measured as NO_2 + NO_3) also showed no seasonal pattern (Fig. 7). The range in annual mean values from the 4 sites was within ± 0.2 μM. Nitrogen values were highest at RFB (mean = .280 μM). All sites had some low values of NO_2 + NO_3 below the sensitivity of the assay.

Biological Variables

Shoot density of *H. wrightii* is shown in Table 1. The RFB site had the highest mean shoot density at 15,800 shoots m^{-2}, MNB the lowest at 5,800 shoots m^{-2}. ULM was quite similar (annual mean = 9,700 shoots m^{-2}) to LCC (annual mean = 9,100 shoots m^{-2}). A two-way ANOVA showed a significant effect on variance in shoot density by date and by site ($P \leq 0.0001$).

The other biological variable, the diatom epiphyte standing crop on *Halodule wrightii* leaves, varied widely among the four sites (Fig. 8). Upper Laguna Madre had the lowest standing crop of diatom epiphytes for 6 of the 10

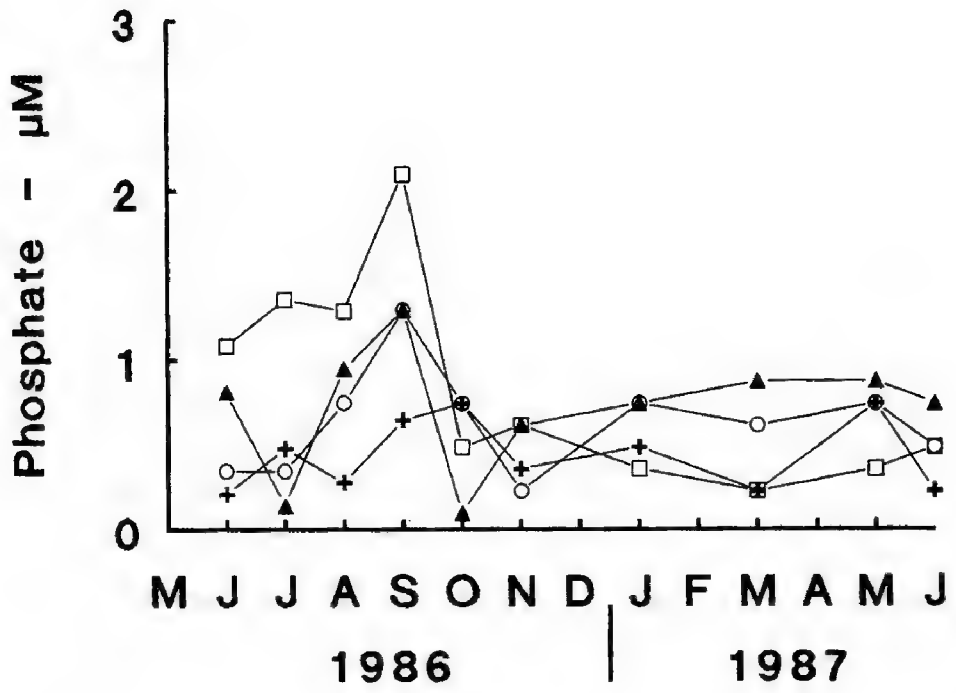


FIG. 6. Measurement of the concentration of dissolved phosphate (as PO_4) at the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). There was not a significant difference in dissolved phosphate between the four sites (ANOVA $p > F = .1730$).

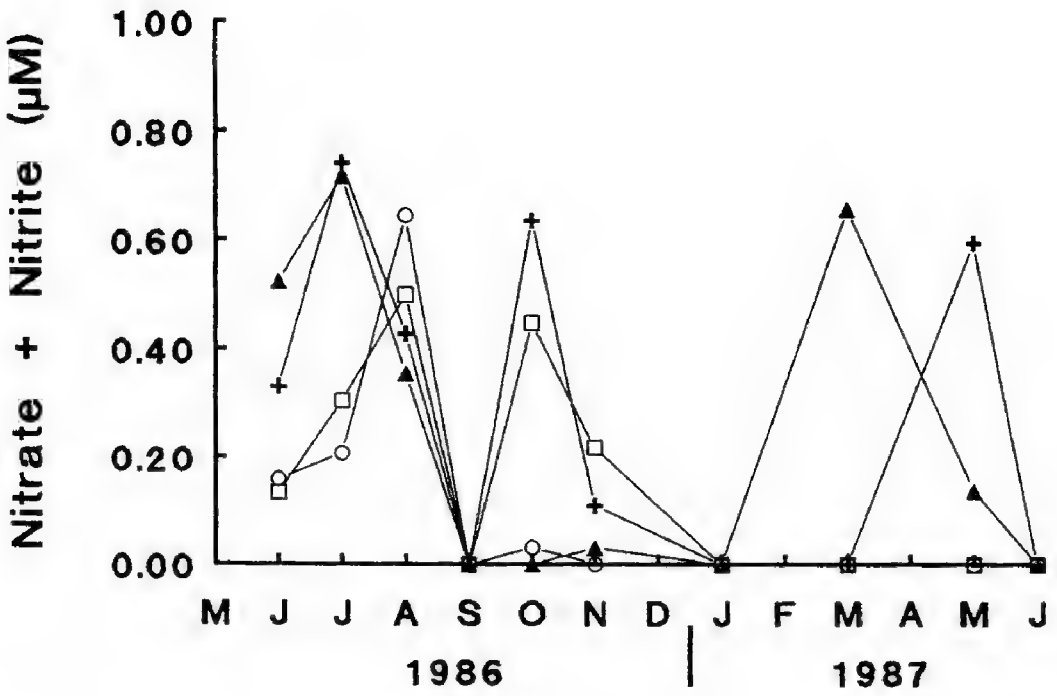


FIG. 7. Measurement of the concentration of dissolved nitrate + nitrite at the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). There was not a significant difference in nitrate + nitrite between the four sites (ANOVA $p > F = .4420$).

TABLE 1
Monthly mean shoot density (per $m^2 \times 10^3$, n = 3) of *H. wrightii* from four South Texas sites for June 1986 to September 1986. The two-way analysis of variance on the effect of date and site on the variability in shoot density of *H. wrightii* did not include June due to missing values.

Site	June 1986	July 1986	August 1986	September 1986	Mean (S.D.)
MNB[1]	9.4	4.2	5.0	4.8	5.8 (2.4)
RFB[2]	*	13.1	12.6	12.7	15.8 (5.0)
ULM[3]	7.2	6.2	16.1	9.2	9.7 (4.5)
LCC[4]	8.1	7.5	11.4	9.5	9.1 (1.7)

Two-way ANOVA

Source	df	SS	F	*P*
Site	3	187.7	10.3	0.010
Date	3	73.4	6.0	0.037

\- = missing data [1] = Mouth of Nueces Bay [2] = Redfish Bay
[3] = Upper Laguna Madre [4] = Lower Corpus Christi Bay

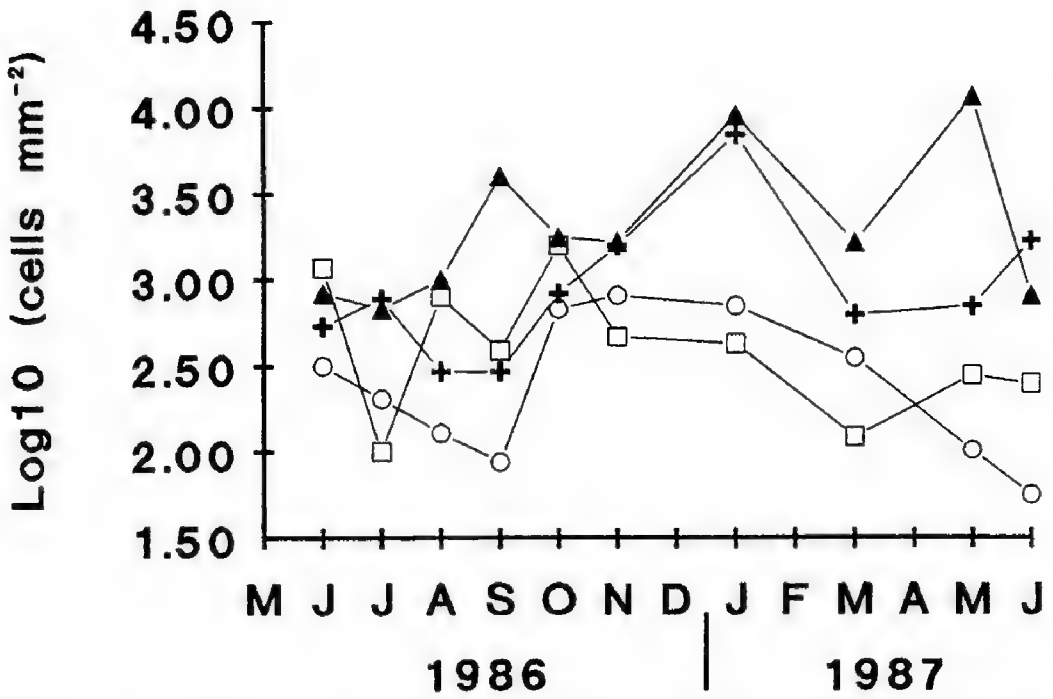


FIG. 8. Measurement of the standing crop of diatoms epiphytic on *Halodule wrightii* from the mouth of Nueces Bay at Indian Point Park (▲), the Upper Laguna Madre at Bird Island Basin (○), Lower Corpus Christi Bay at Fish Pass (□), and Redfish Bay at Fin and Feather (+). Values expressed as $\log_{10}$ of the number of cells mm^{-2} of blade surface. There was a significant difference in diatom epiphyte standing crops between the four sites (ANOVA $p > F = .0238$), with the mean at the Upper Laguna Madre site significantly lower than the mean at the mouth of Nueces Bay site from Scheffe's multiple comparison procedure.

sample periods, MNB the highest for 7 of 10 sample periods. The yearly mean standing crops were quite different between sites with MNB 9 times higher than ULM. There was a negative correlation between standing crop and salinity (salinity × epiphyte, comparing all sites for epiphyte standing crop, $R^2 = -.345$, $P < 0.05$) but salinity was not picked by the stepwise procedure in the multivariate regression (see below).

The epiphytic diatom flora was dominated by pennate forms. *Cocconeis* Ehr. occurred on the blades of *Halodule wrightii* at all sites and was mixed with populations of diatoms in the genera *Rhopalodia* O. Mull., *Nitzschia* Hass., *Navicula* Bory, *Grammatophora*, and *Fragillaria* Lyngb. Taxonomic identification of the diatoms to species level is ongoing and will be reported in a subsequent paper.

Multivariate Analysis

Seven measurements (salinity, temperature, depth, % surface light at the bottom, % surface light at 30 cm, phosphate and inorganic nitrogen) were used as independent variables in a multivariate analysis with diatom standing crop as the dependent variable. Salinity was not singled out in the stepwise multiple variate procedure (Table 2) in either the grouped data set or individual site data sets as a contributing factor to the variability seen in diatom epiphyte standing crops. Seasonality as reflected in temperature and date of collection was important in the grouped analysis as well as singly at ULM ($R^2 = 0.8677$) and RFB ($R^2 = 0.5058$). Light availability (as % slab) was important in the grouped analysis ($R^2 = 0.2396$) and at LCC ($R^2 = 0.5911$). The model generated by the stepwise multivariate regression procedure for all sites and all dates, $y = B_0 + 5.30(\%\ \text{slab}) + 14.80(\%\ \text{sla30}) + 2.81(\text{TEMP}) + 2.19(\text{DEPTH})$, accounted for 52.95% of the variation in the diatom standing crop (from R^2 values) and was statistically significant ($P = 0.0429$).

A separate stepwise regression on the first four sample dates for all sites was run to include seagrass shoot density. Shoot density was not picked as a significant contributor to the variation in epiphyte standing crop values.

DISCUSSION

The water chemistry values for salinity and phosphate reported in this study were similar to previous studies. Salinity has ranged from 12 to 55 ‰ at South Bird Island, and from 10 to 44 ‰ at a southern Corpus Christi Bay station (Brown, Brewton, McGaven, Evans, Fisher and Groat 1976). Measurements of phosphate in the Corpus Christi Bay area have varied from that of Simmons (1957) from Upper Laguna Madre of 0.35 μM to that of Hildebrand and King (1973) of a mean of 2.86 μM from the mouth of the Upper Laguna Madre.

Nitrogen values from previous studies of the Corpus Christi Bay and Upper Laguna Madre show a wide variation. Hildebrand and King (1973) reported

TABLE 2
Stepwise Multiple Variate Procedure

Comparison	Variable	R^2	*P*
All Sites			
Step 1	%SLAB	.2396	.0018
Step 2	%SLA30	.4188	.0001
Step 3	Temperature	.4887	.0382
Step 4	Depth	.5295	.0429
Mouth of Nueces Bay at Indian Point Park			
Step 1	Depth	.2473	.1436
Upper Laguna Madre at Bird Island Basin			
Step 1	Temperature	.5285	.0265
Step 2	Date	.8677	.0078
Lower Corpus Christi Bay at Fish Pass			
Step 1	%SLAB	.5911	.0093
Step 2	NO_2 + NO_3	.7247	.1079
Redfish Bay at Fin and Feather			
Step 1	Temperature	.5058	.0317

nitrogen (nitrate + nitrite) values from a high of 41.3 μM to below detectability from the Laguna Madre (yearly mean = 9.5 μM) and from 79.6 μM to below detectability from Corpus Christi Bay (yearly mean = 20.2 μM). Much lower average values of nitrate + nitrite were reported by Simmons (1957) for the Upper Laguna of 0.109 μM. Although ammonia values were not measured in this study, water column dissolved ammonia values may vary from below 0.1 μM to a high of 50 μM in Corpus Christi Bay and from below 0.1 μM to a high of 10-15 μM in the Laguna Madre with no discernable cyclic pattern (Terry Whitledge, personal communication, 1990). This reported variation in nitrogen values from year to year was likely due to the wide variation in precipitation with virtually no runoff in drought years and periodic inundation during wet years and tropical storms.

Epiphyte density patterns in the Corpus Christi area were similar to that reported by Harlin, *et al.* (1985) for Australia. Standing crop values of epiphytic diatoms in South Texas can be compared to the maximum cell density of epiphytes on artificial substrates in a freshwater reservoir: 4000

cells mm^{-2} (Hoagland, Romer and Rosowski 1982). In the multivariate analysis, the seasonally-related variables of light (reported here as available light, water clarity, and the related parameter of depth) and temperature were an important source of variation to diatom epiphyte standing crops but salinity was not significant. Conversely, a simple linear regression of salinity *vs.* standing crop showed a significant negative correlation. Since ULM is the deepest and clearest site, as well as the saltiest, salinity may be confounded in the multivariate analysis. Also, this study did not include the other major components of the algal epiflora: diatoms may be more plastic in their salinity tolerance than red or green algae. Casual visual examination of the blades showed very few red algal epiphytes at ULM. The salinity "sieve" for diatoms may be at higher concentrations, such as that seen by Kendrick, Walker and McComb (1988) where 50 ‰ was used to delineate between species groupings in cluster analysis of macroalgal epiphytes of *Amphibolis antarctica*. Thus, in a drought year salinity may become highly important to diatom distribution and abundance.

The lack of correlation of water column nutrients, to epiphyte standing crops at 3 of the 4 sites, reflects the idea that in oligotrophic waters, macrophytes and associated epiphytes may depend more upon remineralized nutrients from the sediment pool as has been pointed out by Eminson and Moss (1980). It may also be that the lack of correlation between water column nutrient values and epiphyte standing crop in this study was due to the sample regime: more frequent sampling or continuous measurements of water column nutrients might help define the correlation. With the feasible importance of a sediment nutrient pool in seagrass and epiphyte biology, future studies of the factors affecting epiphyte distribution and abundance should measure the concentration of sediment nutrients. Blade growth differences between the sites may also explain the differences in standing crop between sites. ULM, the site with the highest light values, had the lowest standing crops. The site with the lowest light values, MNB, had the highest standing crops. This inverse relationship between light and epiphyte cover could be due to a reduced blade growth rate of *H. wrightii* at the MNB site, the site with the lowest shoot density. Blade growth rates were not measured in this study and would be a valuable parameter to consider for future studies of interactions between epiphytes and their host macrophytes.

SUMMARY

Four *Halodule wrightii* seagrass beds in South Texas were studied from June 1986 through June 1987. The sites were significantly different in depth, water clarity (as % surface light at 30 cm), salinity, shoot density (of *H. wrightii*) and standing crop of epiphytic diatoms on the blades of *H. wrightii*. Using stepwise multivariate statistical procedures, the main physical/chemical variables which could account for the variation in epiphyte standing

crop on *H. wrightii* blades were temperature and the related variables of light and depth.

ACKNOWLEDGMENTS

This research was conducted with funds from a Lund Fellowship. I thank Warren Pulich, Jr. for his invaluable help setting up the field sites for this project, and Robert Jones, Director, University of Texas Marine Science Institute, for providing laboratory facilities. I would also like to thank the anonymous reviewers for valuable criticism which improved this manuscript. The University of Texas at Austin Marine Science Institute Contribution No. 822.

LITERATURE CITED

BROWN, L.F., J.L. BREWTON, J.H. McGAVEN, T.J. EVANS, W.L. FISHER, and C.G. GROAT. 1976. *Environmental Geologic Atlas, Texas Coastal Zone.* Bureau of Economic Geology, The University of Texas at Austin, Austin, Texas.

EMINSON, D. and B. MOSS. 1980. The composition and ecology of periphyton communities in freshwaters. I. The influence of host type and external environment on community composition. *British Phycological Journal.* 15:429-446.

FREESE, L.R. 1952. Marine diatoms of the Rockport, Texas bay area. *Texas Journal of Science.* 3:333-386.

HARLIN, M.M., Wm.J. WOELKERLING and D.I. WALKER. 1985. Effects of a hypersalinity gradient on epiphytic Corallinaceae (Rhodophyta) in Shark Bay, Western Australia. *Phycologia.* 24:389-402.

HILDEBRAND, H. and D. KING. 1973. A Preliminary biological study of the Cayo del Oso and the Pita Island area of the Laguna Madre: annual report before power plant operation 1972-1973. Central Power and Light Company, Corpus Christi, Texas.

HOAGLAND, K.D., S.C. ROMER and J.R. ROSOWSKI. 1982. Colonization and community structue of two periphyton assemblages, with emphasis on the diatoms (Bacillariophyceae). *American Journal of Botany.* 69:188-213.

KENDRICK, G.A., D.I. WALKER and A.J. McCOMB. 1988. Changes in distribution of macroalgal epiphytes on stems of the seagrass *Amphibolis antarctica* along a salinity gradient in Shark Bay, Western Australia. *Phycologia.* 27:201-208.

MACAULEY, J.M., J.R. CLARK and W.A. PRICE. 1988. Seasonal changes in the standing crop and chlorophyll content of *Thalassia testudinum* Banks ex Konig and its epiphytes in the Northern Gulf of Mexico. *Aquatic Botany.* 31:277-287.

MEDLIN, L.K. 1983. Community analysis of epiphytic diatoms from selected species of macroalgae collected along the Texas coast of the Gulf of Mexico. Ph.D. Dissertation, Texas A & M University, College Station. 150 pp.

MORGAN, J.C. 1975. Studies on the diatoms of the northwestern Gulf of Mexico: identification, distribution, and cultural studies. M.A. Thesis, The University of Texas, Austin. 67 pp.

PARSONS, T.R., Y. MAITA and C.M. LALLI. 1984. *A Manual of Chemical and Biological Methods for Seawater Analysis.* Pergamon Press, Oxford. 173 pp.

ROUND, F.E. 1981. *The Ecology of Algae.* Cambridge University Press, Cambridge. 653 pp.

SAS INSTITUTE, INC. 1987. *SAS User's Guide for Personal Computers: Statistics.* 6th ed. Cary, North Carolina. 956 pp.

SIMMONS, E.G. 1957. An ecological survey of the Upper Laguna Madre of Texas. *Publications of the Institute of Marine Science, The University of Texas.* 4:156-200.

SULLIVAN, M.J. 1979. Diatom communities from a Delaware salt marsh. *Journal of Phycology.* 11:384-390.

WALKER, D.I. 1985. Correlations between salinity and growth of the seagrass *Amphibolis antarctica* (Labill.) Sonder & Aschers., in Shark Bay, Western Australia, using a new method for measuring production rate. *Aquatic Botany*. 23:13-26.

WOOD, E.J.F. 1963. A study of the diatom flora of fresh sediments of the South Texas bays and adjacent waters. *Publications of the Institute of Marine Science, The University of Texas*. 9:237-310.

Contributions in Marine Science. (1991) Vol. 32.

EFFECTS OF THE NUMBER OF SHORT SHOOTS AND PRESENCE OF THE RHIZOME APICAL MERISTEM ON THE SURVIVAL AND GROWTH OF TRANSPLANTED SEAGRASS *THALASSIA TESTUDINUM*

David A. Tomasko[1], Clinton J. Dawes, and Margaret O. Hall[2]

University of South Florida
Department of Biology
Tampa, Florida 33620

ABSTRACT

The number of short shoots per transplant unit of *Thalassia testudinum* had a marked effect on survivorship. Four-shoot units had survival rates over 85% nine-months post-transplantation, two-shoot units averaged 60%, and one-shoot units averaged 33%. Four-shoot units were also more likely to produce new shoots than one- or two-shoot units. The presence or absence of a rhizome apical meristem had no effect on survivorship of transplant units or the probability of surviving units producing new shoots. However, transplant units with intact rhizome apical meristems produced more new short shoots than transplant units without rhizome apicals. The study indicates that survival of bare rhizome sprigs of *Thalassia testudinum* increases with the number of short shoots, and more rapid proliferation of new short shoots occurs in units with intact rhizome apical meristems.

INTRODUCTION

Seagrass meadows provide food and shelter for a majority of commercially and recreationally important fisheries species during at least part of their life cycles (Thayer, Bjorndal, Ogden, Williams and Zieman 1984). In Tampa Bay, Florida, Lewis, Durako, Moffler and Phillips (1985) reported that 81% of the seagrass coverage has been lost either directly by dredging and filling, or indirectly by reduced water clarity. Because of this decline in valuable fisheries habitat, interest in seagrass transplantation has developed.

As *Thalassia testudinum* Banks ex Konig is the dominant seagrass in terms of biomass and coverage in the Gulf of Mexico (Zieman 1982), it has received much attention as a transplant species. Although Thorhaug (1974) had little

---

[1] Present address: Sarasota Bay National Estuary Program, 1550 Ken Thompson Parkway, Sarasota, Florida 34236

[2] Present address: Florida Department of Natural Resources, Marine Research Institute, St. Petersburg, Florida 33701

success replanting *T. testudinum* from vegetative stock, van Breedveld (1975) reported high survival rates using plugs (seagrass plus sediment) in Tampa Bay. The use of fruits and seeds to revegetate barren sites has been advocated (Thorhaug 1974), although high mortality rates and loss of seedlings has been a problem (Phillips 1980; Durako, personal communication, 1988). Low levels of seed production have been reported in Tampa Bay *T. testudinum* populations (Lewis, *et al.* 1985), further illustrating the difficulty of establishing populations from seeds.

Studies on the feasibility of large-scale transplanting projects involving *Thalassia testudinum* would have greater success if rhizome sections of mature plants rather than seeds or seedlings are used. A potential problem in transplanting rhizome sections of *T. testudinum* is a lack of meristematic tissue in plugs or sods removed from a bed to obtain planting units. Rhizome apices are not common; few would be expected to occur in the size plugs or sods typically used in revegetation projects.

Tomlinson (1974) concluded that *Thalassia testudinum* could not be propagated without an active rhizome apex, because production of new short shoots and rhizomes occurs exclusively from rhizome meristematic tissue (Tomlinson and Bailey 1972). Expansion of a *T. testudinum* rhizome without an apical meristem could occur only if a new meristem was produced from the base of an older short shoot, an infrequent event (Phillips 1960; Tomlinson 1974). Preliminary work on transplanting *T. testudinum* by Kelly, Fuss and Hall (1971) and van Breedveld (1975) resulted in the authors reaching opposite conclusions on the need for rhizome apices, perhaps due to inadequate sample sizes in both studies.

Another problem with obtaining transplanting material from plugs or sods could be variability in the number of short shoots on any particular rhizome section. Based on studies concerning terrestrial clonal plants, the number of short shoots on any particular rhizome section may be critical for ameliorating effects of sub-optimal conditions which would likely be present at many transplant sites (Ong and Marshal 1979; Salzman and Parker 1985; Alpert and Mooney 1986; Lau and Young 1988). Shoots in *Thalassia testudinum* are physiologically integrated (Tomasko and Dawes 1989); the collection of shoots along the rhizome, rather than individual shoots, would be the functional individual. Amelioration of localized sub-optimal conditions should occur, and the rhizome sections with only one short shoot would be expected to have higher mortality rates than sections with two or more short shoots.

The purpose of this study was to examine several factors which could affect survival and growth of transplanted *Thalassia testudinum* rhizome sections. The specific questions asked were:

1) Does the presence or absence of a rhizome apical meristem affect survival and expansion of transplant units?
2) Does the number of short shoots on a rhizome section affect survival and expansion of transplant units?

MATERIALS AND METHODS

The donor bed of *Thalassia testudinum* was located 200 m offshore Mullet Key, Florida (27° 37.2′ N, 82° 41.7′ W) at the mouth of Tampa Bay. Planting units were collected June 20, 1988, a time of active seagrass growth (Dawes and Lawrence 1980; Rice, Trocine and Wells 1983).

Bare rhizome sections with intact apical meristems were collected to obtain 30 replicates of each of the following six treatments: rhizome sections with apical meristems containing 1, 2, or 4 short shoots, and rhizome sections without apical meristems containing 1, 2, or 4 short shoots. Rhizome apical meristems were removed by cutting the terminal portion of the rhizome section with a pair of scissors. Collecting rhizome sections with intact apical meristems ensured us that the first short shoot was always the youngest short shoot, short shoot 2 the second youngest, etc. It was important to know relative ages of short shoots, as older short shoots are more likely to produce rhizome apical meristems than younger ones (Tomlinson 1974).

The rhizome sections were transplanted into a bare area surrounded by a *Halodule wrightii* bed offshore Port Manatee, Florida (27° 36.4′ N, 82° 34.2′ W) in 0.5 m (MLW) of water. Wave action is low to moderate at the transplant site, and boat traffic is minimal (personal observation). Rhizome sections were transplanted at 0.5 m centers in a pre-established random design into a 7.0 m by 5.5 m plot. Prior to transplantation, rhizome sections were tagged and their location within the plot, as well as presence or absence of a rhizome apical meristem and the number of shoots present, was recorded.

Transplanted rhizome sections were monitored every three months for a year. Survival and production of new short shoots were recorded for each monitor. Due to high survivorship and proliferation of many transplant units, delimiting individual units became increasingly difficult with time. The nine month post-transplant monitor was the last one where accurate counts of short shoot formation could be determined; statistical analysis of the data involves the nine month monitor. Statistical analysis of the data employed the Chi-square test to determine differences in survivorship and production of new shoots as affected by presence of a rhizome apical meristem and the number of short shoots on the transplant unit.

RESULTS

The number of shoots per transplant unit had a highly significant effect on survivorship, the probability of new shoot formation, and the number of shoots formed per transplant unit (Table 1). Survivorship was highest in four-shoot units at all times, and least in one-shoot units (Fig. 1). Four-shoot units were more likely to produce new short shoots, and would form more short shoots than either two- or one-shoot transplant units (Table 2).

The presence of a rhizome apical meristem had no effect on survivorship of transplant units (Table 1), as shown in Table 2. Rhizome apical meristems also had no significant influence on the probability of transplant units producing new shoots (Table 1). However, the presence of an apical meristem did significantly affect the number of new shoots formed (Table 1). For all three categories of shoot numbers per transplant units, those with rhizome apical meristems produced greater numbers of new shoots per surviving unit than those without rhizome apical meristems (Table 2).

When both survival and proliferation are taken into account, only four-shoot units showed a net increase in the number of short shoots per transplant plot (Fig. 2). Four-shoot units with rhizome apical meristems showed an increase in the total number of shoots of 50%. Four-shoot units without

TABLE 1
Results of Chi-square analysis on effects of the number of short shoots and presence of the rhizome apical meristem on survivorship, production of new shoots, and number of new shoots produced. Data are from nine months post-transplantation.

	Number of of Shoots	**Presence of Rhizome Apical**
Survivorship	15.77*	0.33
New Shoot Production†	17.15*	3.50
Number of New Shoots†	73.09*	36.13*

† includes only those units which had survived
* $P \leq 0.01$

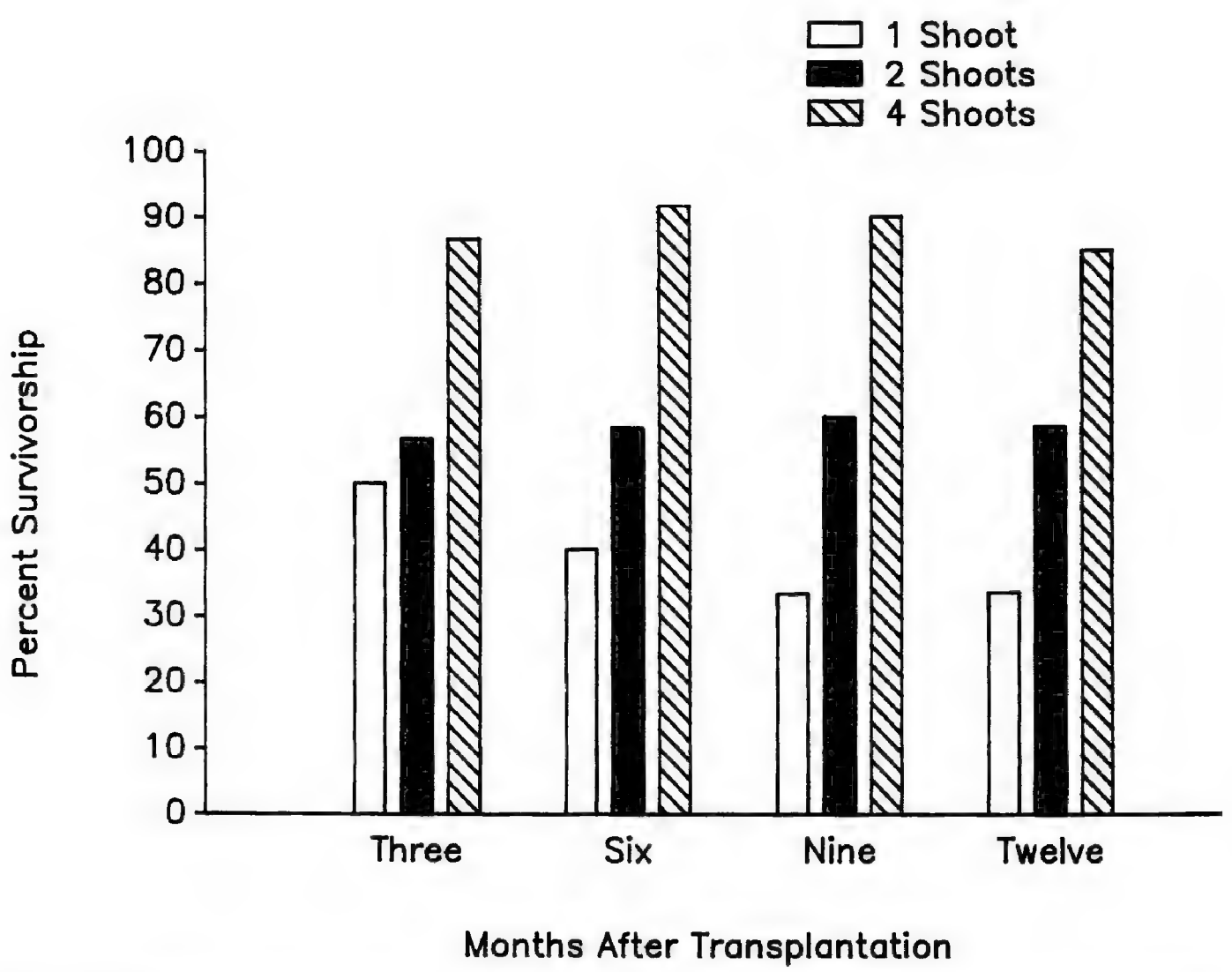


FIG. 1. Percent survivorship of transplant units with 1, 2, or 4 short shoots at 3, 6, 9, and 12 months post-transplantation. As there was no effect of presence of a rhizome apical meristem on survivorship, data are combined for with and without apical meristems.

TABLE 2
Summary of data from nine months post-transplantation. "+" = with intact rhizome apical meristem, "−" = without intact rhizome apical meristem.

	1 Shoot		2 Shoots		4 Shoots	
	+	-	+	-	+	-
Percent Survival	23.3	43.3	60.0	60.0	90.0	90.0
Percent With New Shoots*	57.1	38.5	50.0	27.8	81.5	40.7
Average Number of New Shoots*	0.9	0.4	1.3	0.2	2.5	0.8

*includes only those units which had survived

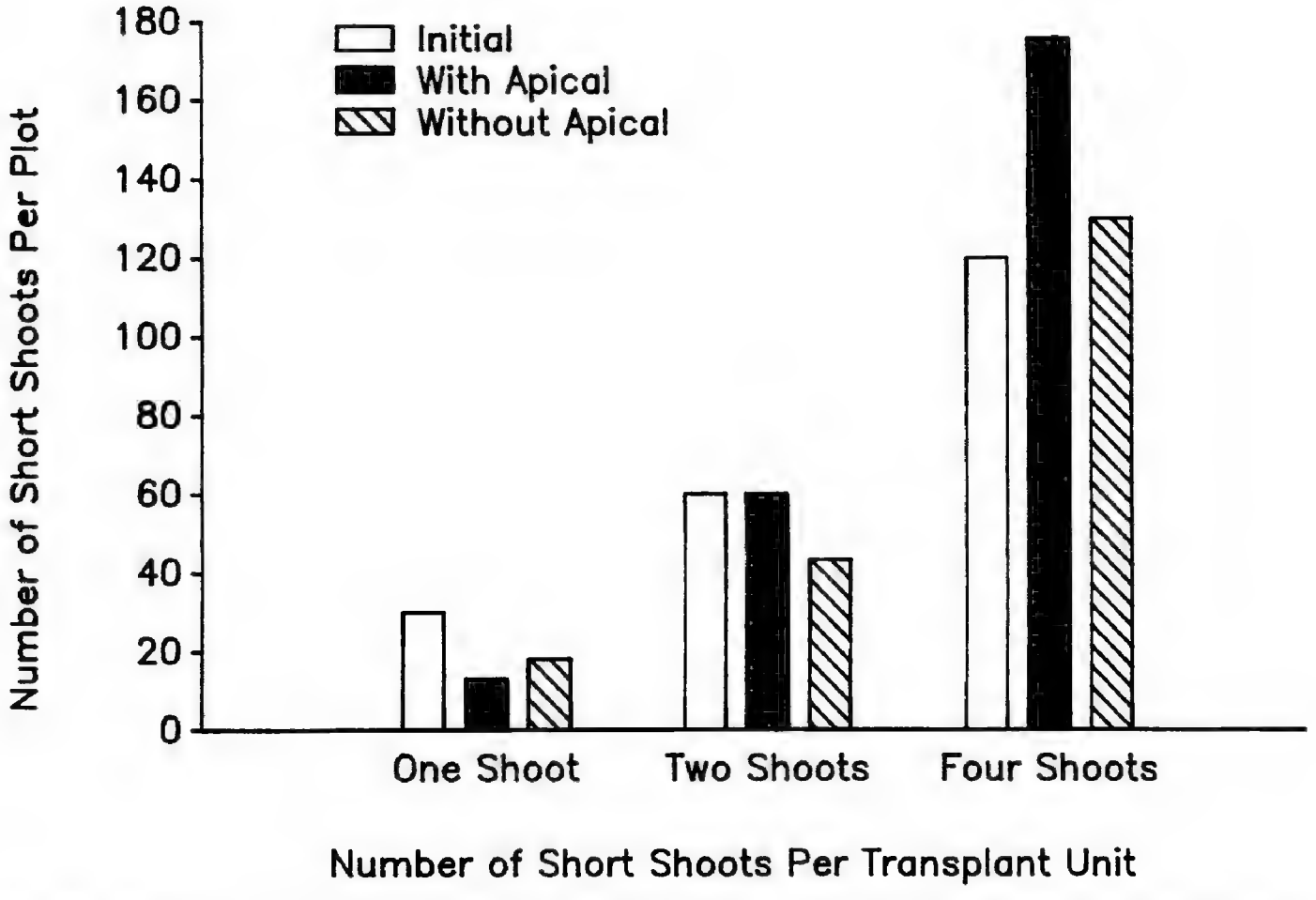


FIG. 2. Number of short shoots initially and nine months post-transplantation for all combinations of shoot number per transplant unit and presence or absence of rhizome apical meristems.

rhizome apical meristems showed a very slight increase. Two-shoot units with rhizome apical meristems showed no net increase. Two-shoot units without rhizome apices, and one-shoot units with and without rhizome apices all showed a decline in total shoot numbers from the initial values.

DISCUSSION

If large-scale revegetation projects involving seagrasses are to continue to draw public funding and enthusiasm, the probability of success must exceed past results. In his review of transplant projects, Lewis (1987) listed several projects involving *Thalassia testudinum* at Craig Key, Florida that varied greatly in survival rates. The number of plots with survival rates over 25% (2) was surpassed by the number of plots with survival rates under 25% (6). Notwithstanding the influences of currents, water clarity and sediment type on transplantation success, it seems that possible biological bases for success or failure have not received adequate attention. With the short shoots along any section of rhizome functioning as a collection of physiologically integrated units (Tomasko and Dawes 1989), clonal growth properties of *Thalassia testudinum* might explain some of the various degrees of success of past transplantation projects.

Tomlinson (1974) stated that *Thalassia* could not be propagated without an active rhizome apex. Our data show that proliferation of short shoots will occur using transplant units without a rhizome apex, but at a much slower rate than in units with intact apices. In addition, the formation of new rhizome apices appears to be greatest in transplant units with greater numbers of shoots, which themselves have higher survival rates.

As illustrated in Figure 2, the best possible combination for transplant projects is that of four-shoot units with intact rhizome apical meristems. The combination of high survivorship (85%) and high rates of new shoot production (average of 2.5 per surviving unit) made further quantitative monitoring of the entire plot impossible nine months after transplantation.

Lewis (1987) pointed out the low survivorship of *Thalassia testudinum* transplant plots using bare rhizome sprigs (the method used here) *versus* plugs. Perhaps the reason for the high survivorship rates here is the size of the rhizome sections used, and the greater number of shoots per transplant unit. The fact that the sprigs had no accompanying sediment seems to be of little consequence. In addition, Fonseca, Thayer and Kenworthy (1987) suggested unvegetated sites are to be avoided as transplant areas, and stressed "compressed succession" as a transplanting technique for *Thalassia testudinum.* However, unexplained gaps in vegetation cover are universal phenomena (Bazzaz 1979). Subtidal areas unvegetated by seagrasses, but of similar sediment texture, and water depth as nearby seagrass meadows, may be more than adequate for seagrass transplant success, as this study, which was done in such an area, suggests.

Present results indicate an important role of the number of short shoots and presence of the rhizome apical meristem on the survivorship and proliferation of transplanted *Thalassia testudinum*. Perhaps variation in success rates of prior transplanting projects involves biological as well as physical processes. Some areas deemed unsuitable for seagrass transplant success based on prior failure might actually be adequate with a modified transplanting technique.

ACKNOWLEDGMENTS

The authors would like to thank Ms. Evamaria W. Koch and Mr. Charles Kovach for assisting in the transplanting and monitoring phases of the study. Ms. Koch was especially helpful in the initial phases of the study. The authors would also like to thank Dr. C. McMillan and Dr. R. Phillips for their reviews of this manuscript. This study was made possible through a grant to one of the authors (CJD) from the Florida Department of Natural Resources Bureau of Marine Research. DNR Contract C4627.

LITERATURE CITED

ALPERT, P. and H.A. MOONEY. 1986. Resource sharing among ramets in the clonal herb, *Fragaria chiloensis*. *Oecologia*. **70**:227-233.

BAZZAZ, F.A. 1979. The physiological ecology of plant succession. *Annual Review of Ecology and Systematics*. **10**:351-371.

DAWES, C.J. and J.M. LAWRENCE. 1980. Seasonal changes in the proximate constituents of the seagrasses *Thalassia testudinum, Halodule wrightii,* and *Syringodium filiforme*. *Aquatic Botany*. 8:371-380.

FONSECA, M.S., G.W. THAYER, and W.J. KENWORTHY. 1987. The use of ecological data in the implementation and management of seagrass restorations. pp. 175-188. In M.J. Durako, R.C. Phillips and R.R. Lewis (eds.), *Proceedings of the Symposium on Subtropical and Tropical Seagrasses of the Southeastern United States.* Florida Department of Natural Resources, Bureau of Marine Resources, Florida Marine Research Publication No. 42.

KELLY, J.A., C.M. FUSS, and J.R. HALL. 1971. The transplanting and survival of turtle grass, *Thalassia testudinum,* in Boca Ciega Bay, Florida. *Fishery Bulletin*. **69**:273-280.

LAU, R.R. and D.R. YOUNG. 1988. Influence of physiological integration on survivorship and water relations in a clonal herb. *Ecology*. **69**:215-219.

LEWIS, R.R. 1987. The restoration and creation of seagrass meadows in the southeastern United States. pp. 153-174. In M.J. Durako, R.C. Phillips and R.R. Lewis (eds.), *Proceedings of the Symposium on Subtropical and Tropical Seagrasses of the Southeastern United States.* Florida Department of Natural Resources, Bureau of Marine Resources, Florida Marine Research Publication No. 42.

_____, M.J. DURAKO, M.D. MOFFLER, and R.C. PHILLIPS. 1985. Seagrass meadows of Tampa Bay — a review. pp. 210-246. In S.F. Treat, J.L. Simon, R.R. Lewis, and R.L. Whitman (eds.), *Proceedings of the Tampa Bay Area Scientific Information Symposium.* Florida Sea Grant Publication No. 65. Burgas Publication Co., Minneapolis, Minnesota.

ONG, C.K. and C. MARSHALL. 1979. The growth and survival of severely shaded tillers in *Lolium perenne* L. *Annals of Botany*. **43**:147-155.

PHILLIPS, R.C. 1960. Observations on the ecology and distribution of the Florida seagrasses. Florida State Board of Conservation Marine Laboratory, Professional Paper Series, No. 2. 72 pp.

_____. 1980. Transplanting methods. pp. 41-56. In R.C. Phillips and C.P. McRoy (eds.), *Handbook of Seagrass Biology*. Garland STPM Press, New York.

RICE, J.D., R.P. TROCINE, and G.N. WELLS. 1983. Factors influencing seagrass ecology in the Indian River Lagoon. *Florida Scientist.* **46**:276-286.

SALZMAN, A.G. and M.A. PARKER. 1985. Neighbors ameliorate local salinity stress for a rhizomatous plant in a heterogeneous environment. *Oecologia.* **65**:273-277.

THAYER, G.W., K.A. BJORNDAL, J.C. OGDEN, S.L. WILLIAMS, and J.C. ZIEMAN. 1984. Role of larger herbivores in seagrass communities. *Estuaries.* 7:351-376.

THORHAUG, A. 1974. Transplantation of the seagrass *Thalassia testudinum* Konig. *Aquaculture.* **4**:177-183.

TOMASKO, D.A. and C.J. DAWES. 1989. Evidence for physiological integration between shaded and unshaded short shoots in *Thalassia testudinum. Marine Ecology — Progress Series.* **54**:299-305.

TOMLINSON, P.B. 1974. Vegetative morphology and meristem dependence — the foundation of productivity in seagrasses. *Aquaculture.* **4**:107-130.

———— and G.W. BAILEY. 1972. Vegetative branching in *Thalassia testudinum* (Hydrocharitaceae): a correction. *Botanical Gazette.* **133**:43-50.

VAN BREEDVELD, J. 1975. Transplanting of seagrasses with emphasis on the importance of substrate. Florida Department of Natural Resources, Marine Research Laboratory, Florida Marine Research Publication, No. 17. 26 pp.

ZIEMAN, J.C. 1982. The ecology of seagrasses of south Florida: a community profile. U.S. Fish and Wildlife Services, Office of Biological Services, Washington, D.C. **FWS/OBS-82/25.** 158 pp.

Contributions in Marine Science. (1991) Vol. 32.

THE EFFECT OF FRESHWATER INFLOW ON MACROBENTHOS IN THE LAVACA RIVER DELTA AND UPPER LAVACA BAY, TEXAS

Richard D. Kalke and Paul A. Montagna

The University of Texas at Austin
Marine Science Institute
P.O. Box 1267
Port Aransas, Texas 78373 USA

ABSTRACT

A two year study on the effects of freshwater inflow on macrobenthos at selected sites in the upper portion of the Lavaca River and Bay was conducted from November 1984 through August 1986. Lowest densities occurred in lakes, and highest densities occurred in creek deltas. Sediment grain size did not have a significant effect on spatial distributions. The first year of the study had higher inflow rates than the second year. Temporal changes of inflow had a larger effect on spatial variability, because freshwater species had extended ranges during the high-flow year. Temporal variation in the benthic community during the study was caused by population changes of low salinity species, and this was associated with high inflow rates during the first year. Freshwater inflow is apparently necessary to induce the recruitment of low salinity species in the upper bay. Chironomid larvae and the polychaete, *Hobsonia florida*, increased in density approximately 4 weeks after an inflow event. In contrast, the mollusks, *Mulinia lateralis* and *Macoma mitchelli*, had increased densities during the low inflow period resulting in high benthic biomass. *Streblospio benedicti*, a surface deposit-filter feeder, and *Mediomastus californiensis*, a burrowing deposit-feeder, were also positively correlated with increasing salinity during low inflow periods. Chlorophyll-*a* concentration in the surface water increased with inflow, indicating primary production was stimulated by inflow, and then decreased with increased salinity and increased filter feeder biomass. Non-molluskan benthic biomass was similar during wet and dry periods, and had a positive relationship with chlorophyll-*a* concentration. The amount, timing, and inter-annual variation of inflow influence both the spatial and temporal abundance and biomass of benthic macrofauna.

INTRODUCTION

The upper Lavaca Bay (Fig. 1), located at latitude 28° 41′ North and longitude 96° 36′ West, is part of one of the seven major estuaries along the Texas coast. Lavaca Bay is shallow with a maximum natural water depth of about 2.4 m and a surface area of about 16,576 ha. The perimeter of the upper bay shoreline is lined with patchy *Spartina alternifora* Lois. The surrounding low salinity marshes adjacent to the river are vegetated mainly with *Juncus*

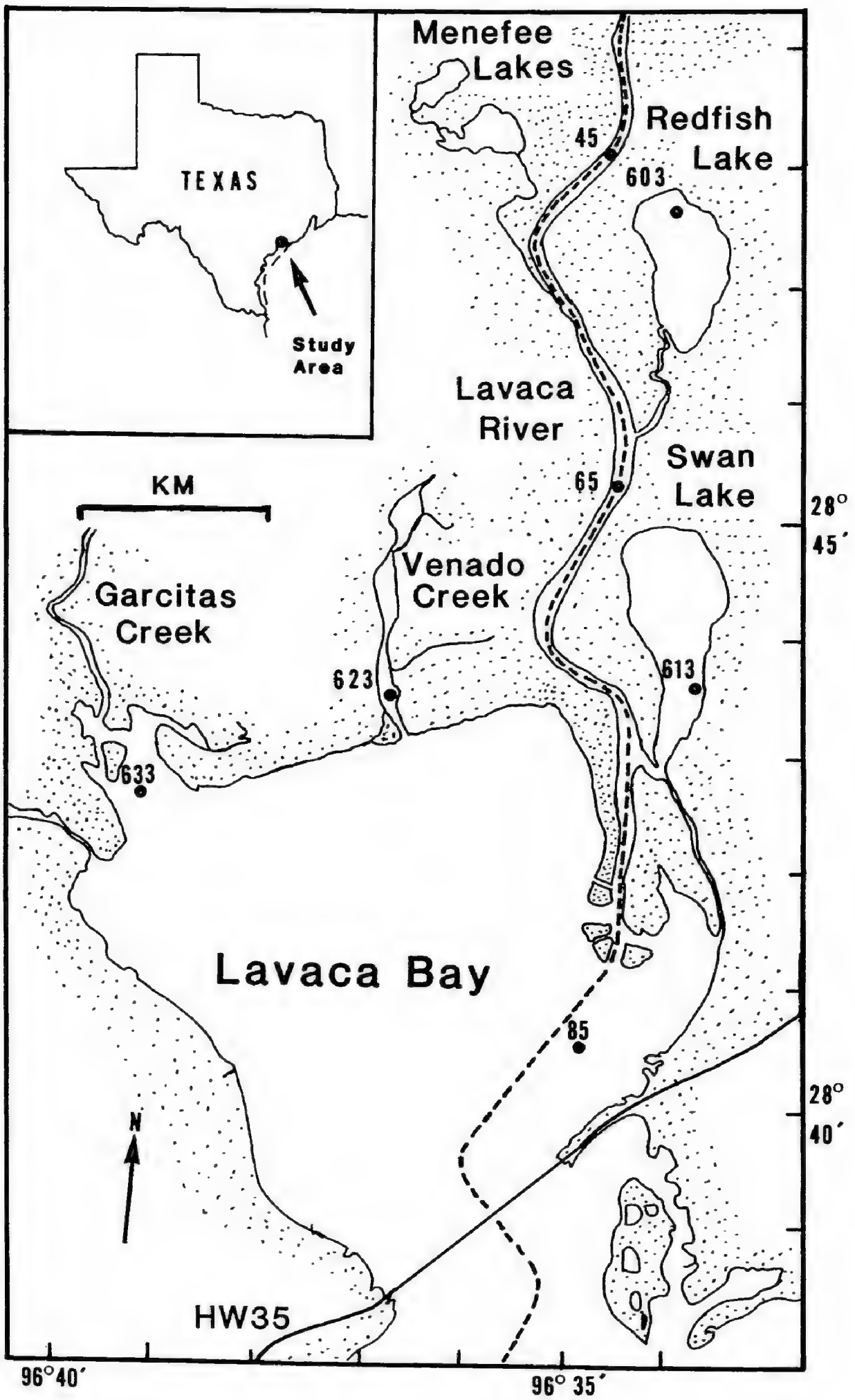


FIG. 1. Location of sample sites in Lavaca River, Delta and Bay.

roemerianus Scheele and *Phragmites communis* Trin. The majority of freshwater inflow into upper Lavaca Bay comes from the Lavaca and Navidad Rivers, while lesser contributions come from Venado, Garcitas and Placedo creeks. The Palmetto Bend Project, a dam, was constructed on the Navidad River in May 1980 to form Lake Texana. This reservoir was constructed to supply water for industrial and municipal use and was not intended for flood control. Major floods are allowed to pass through the flood gates and inundate the marsh system associated with the Lavaca-Navidad River delta. Circulation between the upper and lower bay is modified by the presence of State Highway 35 Causeway, (the remains of the old causeway that has been used as a fishing pier since its destruction by hurricane Carla in 1961) and the Causeway oyster reef which extends across the mouth of the upper bay. The Causeway Reef is a combination of Lap, Hole in the Wall, and Chicken Reefs (Munro 1961). Two tertiary bays or lakes are associated with the Lavaca River. Redfish Lake is approximately 4.8 km and Swan Lake is approximately 1.6 km north of Lavaca Bay (Fig. 1). The total area of Redfish Lake is about 194 ha and Swan Lake is about 259 ha. Both lakes are shallow with a maximum depth of about 1.2 m. The salinity of Redfish Lake is usually similar to the river's while the salinity in Swan Lake is more estuarine due to its proximity and connection with Lavaca Bay *via* Catfish Bayou. Sedimentation from the Lavaca River occurs in upper Lavaca Bay at a rate of 200,000 tons of silt annually. Only minimal amount of fill has occurred at the mouth of the Lavaca River with no appreciable extension of the river delta since 1870 (Shepard 1953).

Benthic studies in early 1900's in the Lavaca-Matagorda Bay area concentrated on oysters to determine their distribution, condition, and the possibility of the development and improvement of the oyster areas (Moore 1907; Moore and Danglade 1915; DeBogert 1913 in Munro 1961; Galtsoff 1931). These oyster bottom studies have described changes in oyster reef distribution over the years which have been attributed to natural and man-induced perturbations. The fluctuation of the area of oyster reefs directly affects the extent of soft bottom benthic communities in the estuary. The Texas Water Development Board (TWDB) was mandated by Senate Bill No. 137 (Schwartz, 64th Legislature, 1975) to conduct studies of the effects of freshwater inflow upon the bays and estuaries of Texas and to estimate the inflows necessary to maintain a suitable ecological environment (TDWR 1982). A number of studies have been conducted in the major estuaries on the Texas coast to measure the effect of freshwater inflow on salinity and nutrient gradients and the abundance and distribution of flora and fauna (Flint, Kalke and Rabalais 1981; Gilmore 1974; Gilmore, Matthews and Clements 1975; Gilmore, Dailey, Garcia, Hannebaum and Means 1976; Harper 1973; Hoese 1960; Holland, Maciolek and Oppenheimer 1973; Holland, Maciolek, Kalke and Oppenheimer 1974; Holland, Maciolek, Kalke, Mullins and Oppenheimer 1975; Jones, Cullen, Lane, Yoon, Rosson, Kalke, Holt and Arnold 1986; Mackin 1971; Matthews, Marcin and Clements 1974; Whitledge, Amos, Benner, Buskey,

Dunton, Holt, Kalke, Montagna, Parker, Stockwell and Yoon 1989). Unlike earlier studies which concentrated on oyster reefs, most of the benthic work in Texas estuaries since the early 1970's have concentrated on the soft bottom benthic communities and their relationship to environmental variables *i.e.*, freshwater inflow, salinity, temperature and sediment type.

The objective of this study was to determine how environmental factors affected by freshwater inflow regulates macrobenthos population abundance, biomass and community structure in the upper Lavaca Bay. Spatial variability was assessed from samples taken in the river, tertiary bays (lakes), and delta. Temporal variability was assessed by sampling monthly and bimonthly over a two-year period. Biological responses were compared with salinity, river streamflow rates, and chlorophyll biomass to assess benthic response to freshwater inflow.

METHODS

The sampling sites included stations 45 and 65 (river sites), 603 and 613 (lake sites), 85 (river delta) and 623 and 633 (creek sites) (Fig. 1). The station locations and numerical designations were established by the TWDB (TDWR 1980). The Palmetto Dam is approximately 8.8 km above the up-river sampling site (Stn. 45) and approximately 24 km from Lavaca Bay.

Fourteen sampling trips were conducted during November, 1984, January, March, April, May, June, July, August, October and December, 1985 and February, March, June and August 1986. Year-1 of the study was from November 1984 through August 1985 and Year-2 was from October 1985 through August 1986.

Benthic core sampling was accomplished using SCUBA or snorkeling. Triplicate samples were collected at each site with 7.5 cm diameter, 30 cm long aluminum cores. The cores were sectioned horizontally by depth at 0-3, 3-10, and 10-20 cm. Sections were placed in a 1 liter jar and preserved with 10% formalin in seawater stained with rose bengal. In the lab, each sediment section was sieved through 0.5 mm mesh and the retained organisms were identified and counted. Wet weight biomass was measured on the numerically dominant individuals and on the remainder of the sample. All mollusk biomass data includes shell weight.

Sediment grain size analysis was performed using standard geologic procedures (Folk 1964; E.W. Behrens, personal communication). Percent contribution by weight was measured for four components: rubble (*e.g.* shell hash), sand, silt, and clay. A 20 cm^3 sediment sample was mixed with 50 ml of hydrogen peroxide and 75 ml of deionized water to digest organic material in the sample. The sample was wet sieved through a 62 μm mesh stainless steel screen using a vacuum pump and a Millipore Hydrosol SST filter holder to separate rubble and sand from silt and clay. After drying, the rubble and sand were separated on a 125 μm screen. The silt and clay fractions were measured using pipette analysis.

Inflow data was obtained from the TWDB. Gauged streamflow data is from a U.S. Geological Survey (USGS) gauge located near Edna, Texas on the Lavaca River. The effect of freshwater inflow of Lavaca Bay benthos was determined by correlating bay-wide abundance and biomass with physical parameters, *i.e.*, salinity, temperature, dissolved oxygen, chlorophyll-*a*, sediment grain size and streamflow. Lavaca River gauged inflow rates, averaged for 14 days prior to and including the first sampling day of each trip, gave the best correlation with mean salinity (Pearson correlation coefficients, $r = -0.55$; $P \leq 0.05$) and was used to assess the impact of recent flow regimes on the fauna. Lagged effects of flow were also examined. For example, salinity changes resulting from freshwater inflow can happen immediately, but effects on benthic reproduction and migration lags behind such changes. Therefore, a flow event may not result in measurable short term changes in the benthic community, but might be detectable later. Pearson correlation

coefficients (r) were calculated for benthos with salinity, temperature, mean 14-day flow and the mean 14-day lag flow. The lag flow was achieved by shifting the flow data forward by one sampling trip. Since 50 correlation tests were performed, it is prudent to take the Bonferroni approach and reject at $P \leq 0.001$ (*i.e.*, 0.05/50).

RESULTS

Historically, upper Lavaca Bay has been mainly supplied with freshwater from the Lavaca and Navidad Rivers. The 49-year daily flow average (1939-1987) for the Lavaca River is 9.4 m^3/s and the forty-year daily flow average for the Navidad River is 16 m^3/s (USGS 1980; Buckner, Carillo and Davidson 1987). Freshwater streamflow rates of the Lavaca River near Edna, Texas indicates that the average daily flow rate for Year 1 of this study was 10 m^3/s, 50% higher than the daily average of 5 m^3/s during Year 2. Since the closing of the dam on the Navidad River in May, 1980 the freshwater inflow pattern has been altered, although it has not deviated much from the historic flow rate of 16 m^3/s. Negligible input from the Navidad River occurred during initial filling of Lake Texana from May 1980 through December 1982. Freshwater releases beginning in December 1982 through December 1983 averaged approximately 35 $m^3/s/d$ on a monthly basis, which is above the 49-year average. The average monthly total inflow from January 1983 through 1986 demonstrates interannual cyclic inflow patterns (Fig. 2). Total inflow is the

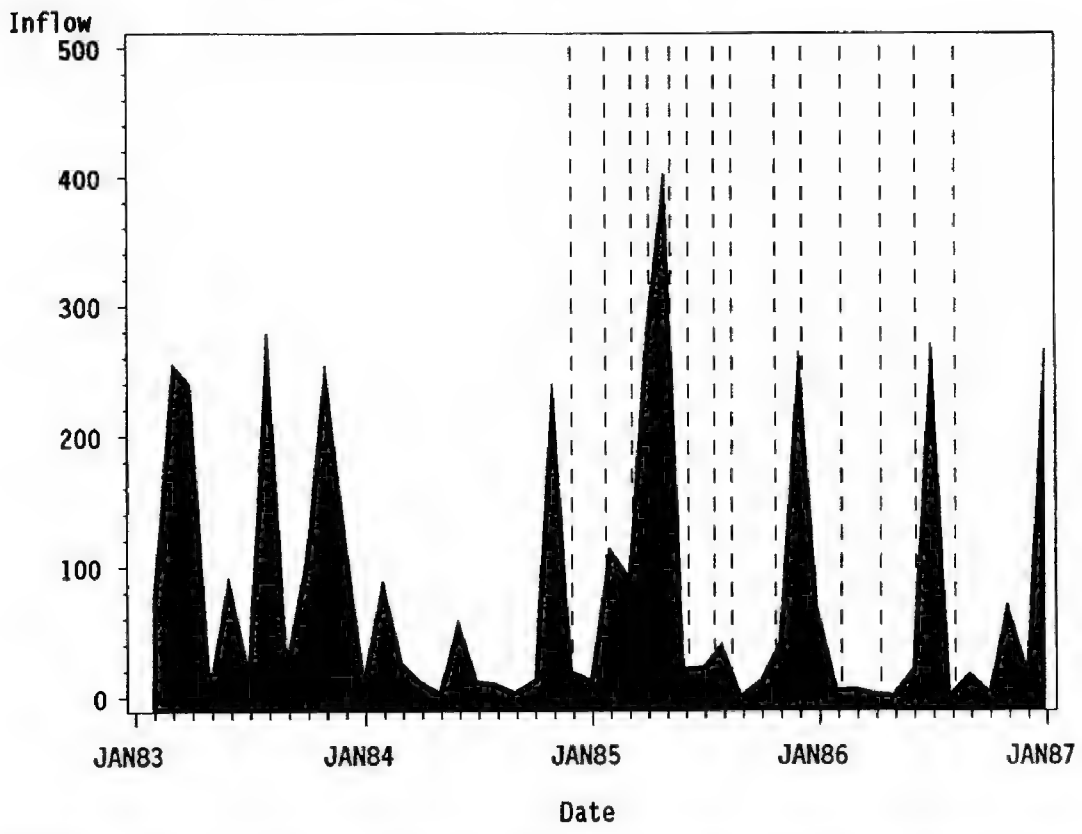


FIG. 2. Total freshwater monthly inflow into Lavaca Bay. Total inflow for each month from 1983-1986 is reported. Dashed vertical lines indicates sampling trips during this study.

sum of gauged inflow, modeled runoff, precipitation and return flows. A wet cycle occurred in 1983, followed by a dry year in 1984, prior to this study. Year-1 of this study was another wet year. Annual variation in total freshwater inflow can be demonstrated by comparing total inflow for September 1983 (722×10^6 m^3/mo), 1984 (1263×10^6 m^3/mo) and 1985 (722×10^6 m^3/mo). Low inflow in September 1983 and 1985 corresponds to dry years and high inflow in September 1984 corresponds to a wet year. Freshwater inflow declined in 1985-86 which resulted in a dry period during Year-2.

We compared stream inflow with salinity measured during sampling trips. We found that salinity correlated best with the average daily stream flow rates for 14-days prior to sampling (Fig. 3). We present this average 14-day inflow rate in Figs. 4-12 to compare inflow with benthic community response.

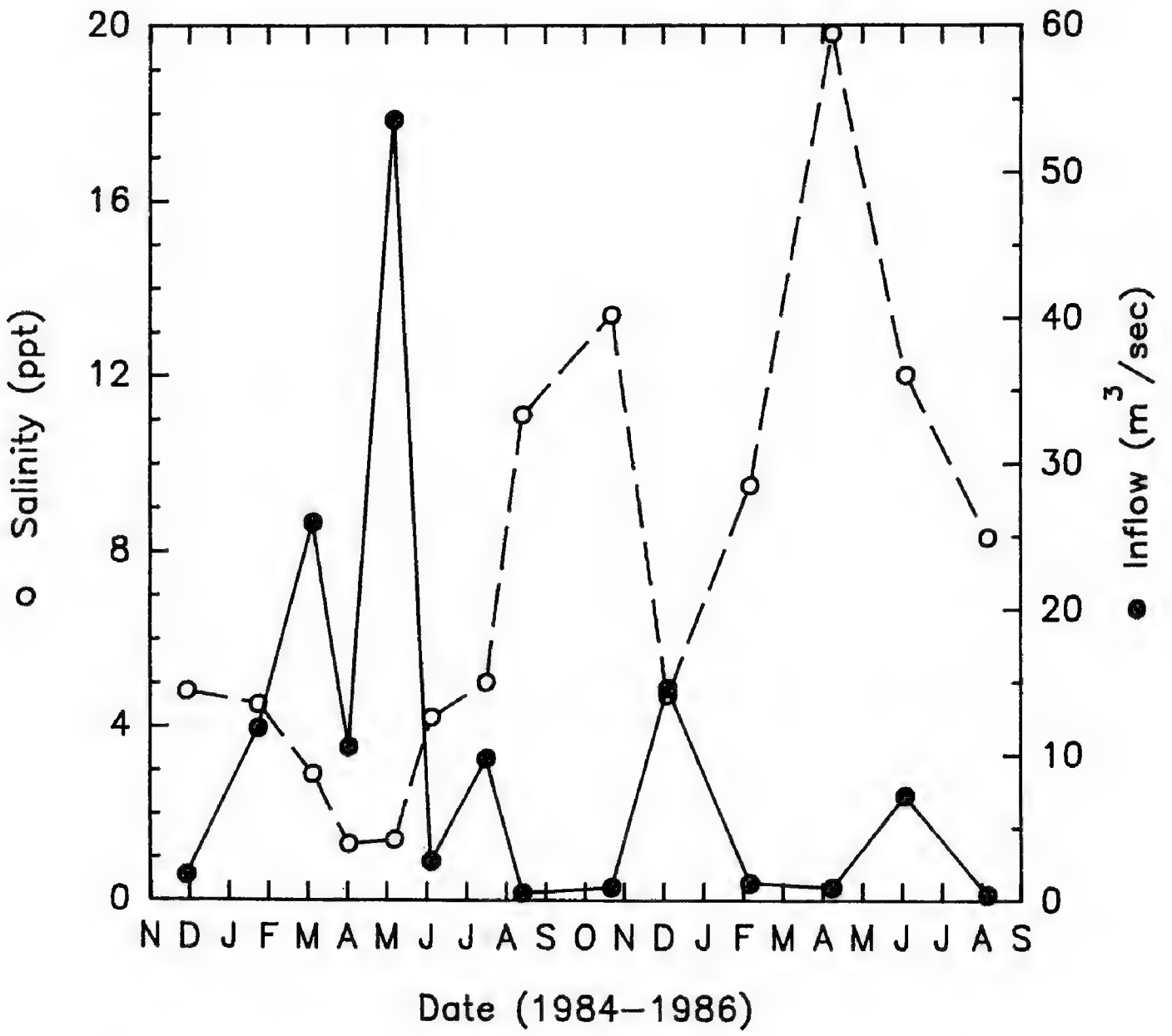


FIG. 3. Lavaca River streamflow 14-day average prior to each sampling trip and its relation to average salinity (dashed line) for all stations by month from November 1984 through August 1986. The 14-day average streamflows (solid line) corresponding with each sampling trip starting in October 1984 are as follows: 2, 12, 26, 11, 54, 3, 10, 0.5, 1, 15, 1, 1, 7, and 0.4 m^3/sec.

January 1984 through December 1985 was a drier period with sporadic daily discharges in January, May, and October 1984 averaging 9.5 m^3/s. From January 1985 through December 1985 increased inflow was noted with releases occurring every month except May, August and September 1985. The daily average flow rate for this period was 18.5 m^3/s. Flow rates were down from January 1986 through December 1986 with releases only in May, June and September, 1986, resulting in a daily average of 8 m^3/s.

The benthic community in upper Lavaca Bay during this study was typical of a low to moderate salinity estuary with only a few dominant species. The dominant species of the 52 taxa collected included the polychaetes, *Mediomastus californiensis, Streblospio benedicti, Laeonereis culveri,* and *Hobsonia florida,* oligochaetes, chironomid midge fly larvae, and the mollusks *Macoma mitchelli* and *Mulinia lateralis* whose combined total abundance comprised 90% of the total taxa abundance (Table 1). Chironomid larvae and *H. florida* were found at low salinity stations. The other dominant taxa: *M. californiensis, S. benedicti,* and *L. culveri* are estuarine species.

The overall average benthic standing crop was not significantly different between Year-1 (5, 505/m^2) and Year-2 (6, 959/m^2), (Table 2). The abundance of benthic fauna was highest in the winter and spring and lowest in the summer and fall (Fig. 4). No significant differences were found with faunal abundance among stations.

Stratification of both infauna abundance and biomass by sediment depth was similar at all stations. The highest concentration of organisms was found in the upper 3 centimeters and decreased with depth to the lowest abundance at the 10-20 cm sediment depth (Table 3). Average abundance by sediment depth was similar for both years at all stations. Distribution patterns of benthic biomass increased with sediment depth, which is opposite of abundance patterns (Table 3). The largest biomass occurred at the 10-20 cm depth zone.

Sediment composition was similar during all three sampling times, but there were station and vertical differences (Table 4). The uppermost river station (45) contained a lot of rubble, as did the surface sediment of the lower river station (65). The river stations were also sandy. Bay and Lake stations had finer sediments.

Biomass increases occurred from March through June 1985, in August 1985 and from February through August 1986 with the highest biomass occurring in Year-2 (Fig. 5). Benthic biomass was positively correlated with salinity, $r = 0.71$ (Table 5). Molluskan biomass dominated the total benthic biomass, so trends of total biomass were strongly influenced by the mollusks (Fig. 5). Mollusk biomass and total benthic biomass were correlated with salinity, $r = 0.73$ (Table 5).

Non-molluskan benthic biomass was comprised of mainly polychaetes, chironomids, rhynchocoels, and a few crustaceans (Table 6). These taxa, as a group, were not significantly correlated with salinity or freshwater inflow (Table 6). However, the main factors influencing total abundance and

TABLE 1
Lavaca Bay species distribution and *n* density/m^2 to a sediment depth of 20 cm by station from November, 1984 - August, 1985.

	Station						
Taxa	**45**	**603**	**65**	**613**	**85**	**623**	**633**
Platyhelminthes							
Turbellaria	0	0	0	0	0	897	0
Rhynchocoela	193	111	244	61	66	83	32
Mollusca							
Gastropoda							
Pyramidellidae							
Odostomia laevigata	0	6	61	11	0	6	26
Odostomia cf. gibbosa	0	6	6	0	11	0	13
Acteonidae							
Acteon punctostriatus	0	0	11	0	0	0	0
Pelecypoda	0	0	0	0	0	0	6
Solenidae							
Ensis minor	0	0	0	0	11	0	6
Tellinidae							
Macoma mitchelli	0	0	249	504	476	210	446
Macoma tenta	0	0	0	0	6	6	0
Mactridae							
Mulinia lateralis	28	6	94	44	194	55	355
Rangia cuneata	0	0	0	0	0	6	0
Solecurtidae							
Congeria leucophaeta	0	0	6	0	0	0	0
Tagelus plebeius	0	0	6	0	50	17	19
Annelida							
Polychaeta							
Phyllodicidae							
Eteone heteropoda	0	0	0	0	11	0	0
Pilargiidae	0	0	6	0	0	0	0
Sigambra tentaculata	0	0	0	0	17	0	0
Ancistrosyllis jonesi	0	0	0	0	11	17	0
Parandalia sp.	0	0	17	0	0	0	6
Hesionidae							
Gyptis vittata	0	0	6	28	11	6	0
Nereiidae	0	0	39	28	0	44	6
Laeonereis culveri	221	6	199	28	17	172	116
Neanthes succinea	0	0	33	0	0	0	0
Gonradidase							
Gycinde solitaria	0	6	6	0	78	17	90
Spionidae							
Polydora socialis	6	0	11	0	50	360	39
Streblospio benedicti	304	1954	1777	1854	1434	3122	1356
Scolelepis texana	0	0	0	0	6	0	0
Cossuridae							
Cossura delta	0	0	0	0	6	0	0
Orbinidae							
Haploscoloplos foliosus	0	0	0	0	6	0	0

	Station						
Taxa	**45**	**603**	**65**	**613**	**85**	**623**	**633**
Capitellidae							
Capitella capitata	0	0	22	39	55	55	19
Mediomastus californiensis	166	127	3089	620	4489	3875	2474
Heteromastus filiformis	0	0	0	0	0	11	0
Ampharetidae							
Hobsonia florida	277	61	61	28	55	210	32
Oligochaeta	769	587	603	277	66	642	349
Hirudinea	11	0	0	0	0	11	0
Crustacea							
Copepoda							
Cyclopoida	0	0	0	0	0	22	0
Hemicyclops sp.	0	0	22	0	0	0	226
Malacostraca							
Mysidacea							
Mysidopsis sp.	0	6	22	11	6	0	0
Mysidopsis almyra	0	17	22	0	0	0	0
Cumacea	0	0	0	0	0	6	0
Cyclaspis varians	0	0	6	0	17	0	6
Oxyurostylis smithi	0	0	5	0	0	0	0
Tanaidacea							
Leptochelia rapax	0	0	6	0	0	0	0
Isopoda							
Edotea montosa	6	0	17	0	28	6	0
Amphipoda							
Ampelisca abdita	0	0	0	6	44	0	0
Monoculodes sp.	0	0	0	11	17	28	78
Corophium louisianum	44	17	426	6	22	6	6
Microprotopus spp.	6	0	0	0	0	0	0
Decapoda							
Callianassa sp. juvenile	0	0	6	0	6	0	0
Callianassa latispina	0	0	17	0	0	0	0
Callinectes sapidus	0	6	0	0	0	0	0
Insecta							
Insecta larvae	0	0	6	0	0	0	0
Diptera							
Chironomidae	2762	614	548	288	22	194	90
Chironomid pupae	22	0	6	0	0	6	0
$\bar{x}$ Total n/m^2	4816	3526	7645	3842	7285	10086	5800

biomass were freshwater inflow and salinity (Table 6). There is a seasonal trend of increasing weight of mollusks in spring and summer. Few relationships with sediment grain size were found. Non-mollusk biomass was weakly correlated with increasing sand content and chlorophyll-*a* concentration.

TABLE 2

Benthos standing crop (n/m^2) to a depth of 20 cm at each station by sampling trip. Mean of 3 replicates.

Station	Month (1984-1986) Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.
45	-	1320	-	10400	-	5120	6280	1470	10500	6120	3640
65	-	4340	-	4570	-	6820	5580	7910	9530	7750	5740
85	-	8760	-	8680	-	7830	6280	7750	8290	7670	6970
603	-	6050	-	2640	-	3640	7980	2480	5500	1400	1160
613	-	388	-	1090	-	2790	1630	1860	4340	4960	2950
623	-	5270	-	8060	-	13600	7980	11300	7750	6900	8530
633	-	-	-	-	-	5810	5350	1860	2250	1320	2870
45	1550	-	1090	-	6200	-	6050	-	22790	-	4880
65	5890	-	6280	-	11900	-	15100	-	12800	-	2790
85	3640	-	2170	-	6200	-	15000	-	7830	-	6000
603	1470	-	310	-	3100	-	7440	-	2170	-	930
613	2090	-	2090	-	11800	-	7980	-	7050	-	2790
623	18100	-	9300	-	17400	-	10200	-	9690	-	7051
633	698	-	5580	-	16000	-	14300	-	11000	-	2480

- station not sampled.

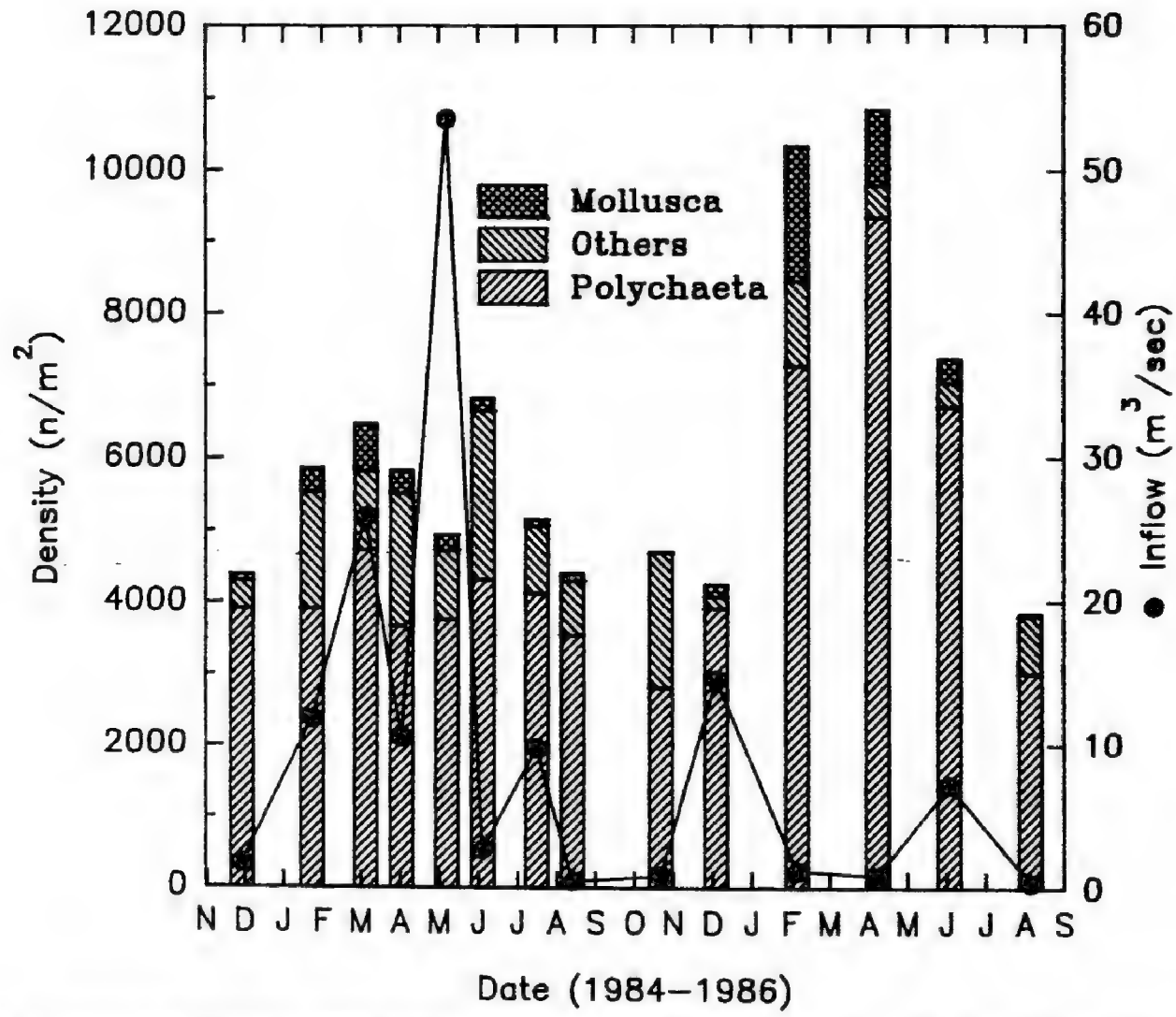


FIG. 4. Benthos standing crop average abundance (n/m^2) for all mollusks, polychaeta, and other taxa (bars) to a depth of 20 cm *vs.* 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

TABLE 3
Vertical distribution of macrofauna. Mean percent occurrence in each core for all stations and sampling trips.

Section	Abundance	Biomass
0 - 3 cm	70%	16%
3 - 10 cm	27%	31%
10 - 20 cm	3%	53%

TABLE 4
Lavaca Bay sediment grain size analysis (% composition). Mean and standard deviation (in parentheses) from 1 replicate taken in November 1984, August 1985, and August 1986.

Station	Depth (cm)	% Rubble	% Sand	% Silt	% Clay
45	0-3	23.0 (31.7)	68.9 (26.6)	3.3 (2.5)	4.7 (2.8)
45	3-10	20.9 (31.3)	70.8 (26.9)	4.1 (5.1)	4.2 (4.4)
45	10-20	29.2 (34.4)	56.8 (29.6)	6.3 (7.0)	7.6 (8.4)
65	0-3	17.9 (19.2)	67.2 (17.6)	6.8 (2.6)	8.1 (2.8)
65	3-10	8.1 (3.8)	47.1 (28.7)	16.1 (8.1)	28.6 (19.6)
65	10-20	2.7 (1.6)	24.0 (10.8)	22.7 (5.2)	50.6 (9.5)
85	0-3	4.0 (5.3)	65.1 (3.5)	13.2 (6.1)	17.6 (2.4)
85	3-10	1.2 (0.6)	59.4 (9.3)	20.0 (8.5)	19.4 (1.5)
85	10-20	0.9 (0.3)	38.8 (17.5)	25.4 (6.2)	34.9 (11.9)
603	0-3	3.4 (2.2)	43.0 (16.2)	14.5 (7.6)	39.2 (17.2)
603	3-10	2.6 (1.2)	38.8 (27.9)	11.2 (1.0)	47.4 (28.0)
603	10-20	2.4 (1.1)	33.9 (28.0)	11.1 (0.9)	52.6 (29.5)
613	0-3	1.3 (1.0)	39.5 (8.9)	19.8 (3.3)	39.4 (12.2)
613	3-10	3.1 (3.9)	47.9 (4.3)	15.7 (3.3)	33.3 (2.3)
613	10-20	0.9 (0.3)	54.5 (7.5)	15.0 (1.9)	29.6 (5.4)
623	0-3	0.5 (0.2)	67.6 (10.3)	16.7 (8.5)	15.2 (3.2)
623	3-10	1.1 (1.3)	62.8 (1.6)	18.5 (0.5)	17.7 (1.9)
623	10-20	0.9 (0.6)	52.8 (6.2)	24.0 (3.3)	22.3 (9.7)
633	0-3	0.4 (0.2)	72.6 (24.6)	14.2 (13.1)	12.8 (11.6)
633	3-10	30.5 (42.2)	42.2 (59.5)	14.6 (9.2)	12.6 (8.1)
633	10-20	0.8 (0.2)	71.5 (12.3)	16.3 (8.5)	11.4 (3.5)

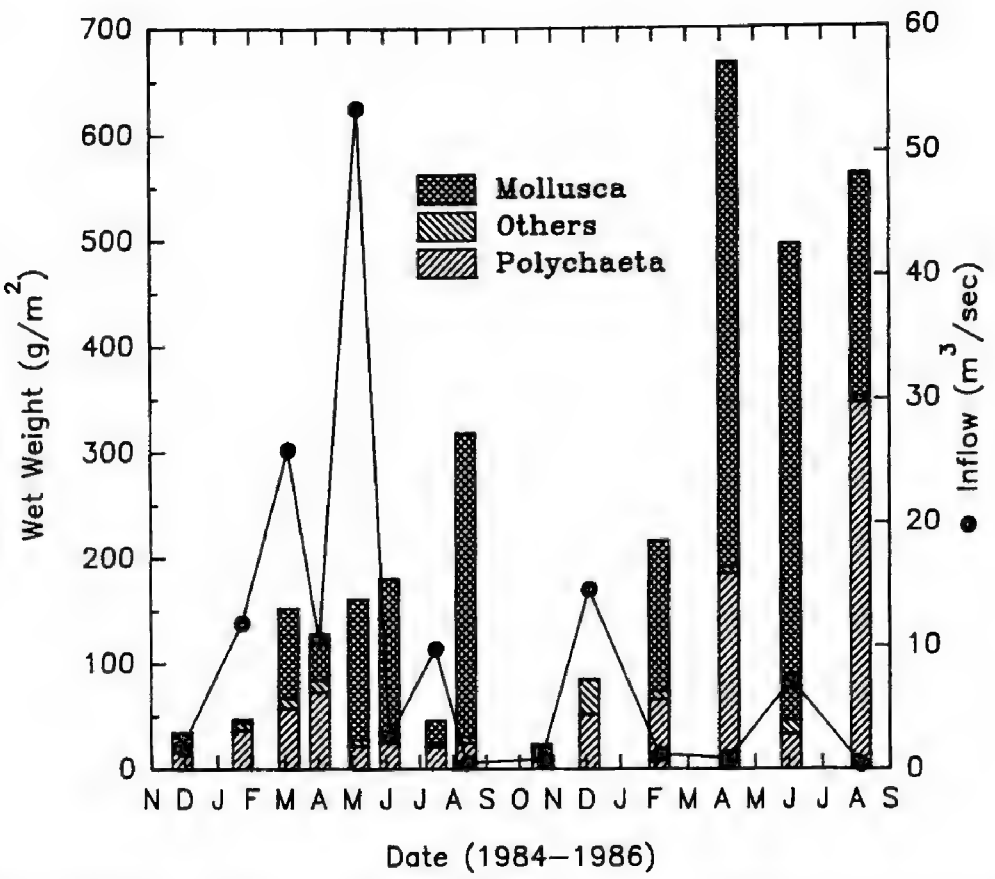


FIG. 5. Benthos average biomass (g/m^2) for all mollusca, polychaeta, and other taxa (bars) to a depth of 20 cm *vs.* 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

TABLE 5

Pearson correlation coefficients (r values) for Lavaca Bay benthos monthly means for all stations and sampling trips, r values without - indicates the correlation is not significant.

	Lavaca River $\bar{x}$ 14 day Stream Flow	$\bar{x}$ 14 day Lag Flow	$\bar{x}$ Salinity by Trip	$\bar{x}$ Temp. by Trip
Lag Flow	-0.05			
$\bar{x}$ Salinity	-0.55*	-0.39		
$\bar{x}$ Temperature	-0.14	0.18	0.35	
Standing crop	-0.18	0.11	0.51	0.11
Total Biomass	-0.19	-0.12	0.71**	0.41
Mollusk Biomass	-0.21	-0.11	-0.73**	0.47
Total Biomass minus Mollusk Biomass	0.10	-0.06	-0.10	-0.44
Chironomid Larvae	-0.03	0.84***	-0.48	0.08
Streblospio benedicti	-0.36	-0.39	0.75**	0.03
Mediomastus californiensis	-0.08	-0.15	0.56*	0.04
Mulinia lateralis	-0.09	0.04	0.34	-0.08
Macoma mitchelli	-0.12	-0.02	0.26	-0.21
Hobsonia florida	0.28	0.79***	-0.47	0.14
Laeonereis culveri	-0.30	-0.44	0.82**	0.27

- $.05 \leq P < 0.01$
-- $0.01 \leq P < 0.001$
--- $0.001 \leq P < 0.0001$

TABLE 6
Correlation between macrofauna and environmental factors. Chlorophyll-*a* data courtesy of John Cullen (in Jones, *et al.* 1987). Based on 98 sample pairs.

	Salinity	Temp.	D.O.	Chl-*a*	Inflow	Sand
Wet weight						
Mollusks	0.39****	0.24*	-	-	-	-
Others	-	0.28**	-	0.21*	-	0.23*
Total Net Weight	0.39****	0.21*	-	-	-	-
Abundance	0.20***	-	-	-	-	-
Chlorophyll-*a*	- 0.30**	0	0.22	-	0.29	-

- not significant
- $0.05 \leq P < 0.01$
-- $0.01 \leq P < 0.001$
--- $0.001 \leq P < 0.0001$
---- 0.0001

Chironomid midge fly larvae and *H. florida* had a positive response to freshwater inflow (Table 5, Figs. 6 and 7). When their abundance was correlated with streamflow, there was a strong lag response the month following an inflow event for chironomid larvae (r = 0.84) and *H. florida* (r = 0.79) (Table 5). Abundance distributions indicated that both species were surface dwellers in the upper 0-3 cm of sediment (Figs. 6 and 7). Their distribution was mainly restricted to the Lavaca River: stations 45 and 65, lakes: stations 603 and 613 and stations 623 and 633 which were associated with freshwater inflow from Venado and Garcitas Creeks (Table 1). High inflows during Year-1 extended the distribution of chironomid larvae (April-May, 1985) and *H. florida* (April-June 1985) into Lavaca Bay to station 85. Higher numbers and extended distributions in Year-1 were a result of high inflow during this period.

The polychaetes, *S. benedicti, M. californiensis* and *L. culveri* all increased in density during low inflow periods. (Figs. 8, 9 and 10). The lowest abundance of *S. benedicti* was during high inflow events and at station 45 upriver throughout the study. The abundance of *S. benedicti* was significantly correlated with salinity (r = 0.75, Table 5). *S. benedicti* was collected mainly from the surface sediments with only a few incidental occurrences in the 10-20 cm core sections (Fig. 8).

Mediomastus californiensis is a burrowing polychaete distributed through the sediment to at least a depth of 20 cm (Fig. 9). Highest densities were found in core sections 0-3 and 3-10 cm and declined below 10 cm. The abundance of *M. californiensis* was significantly correlated with salinity (r = 0.56, Table 5).

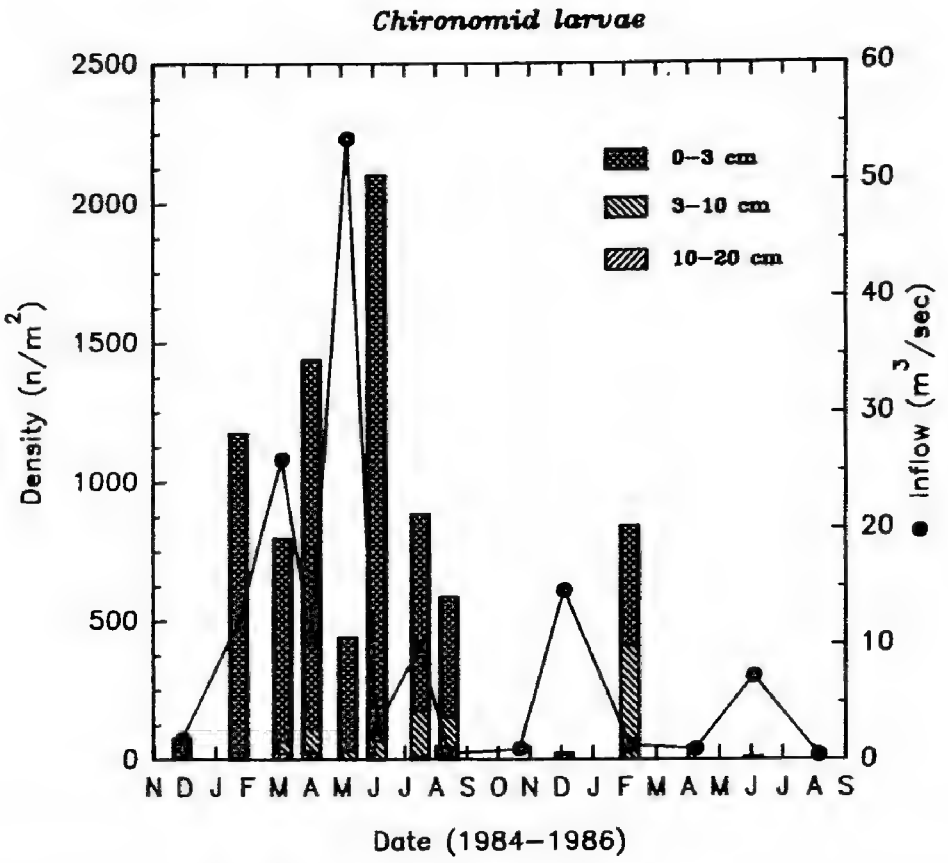


FIG. 6. Chironomid larvae average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

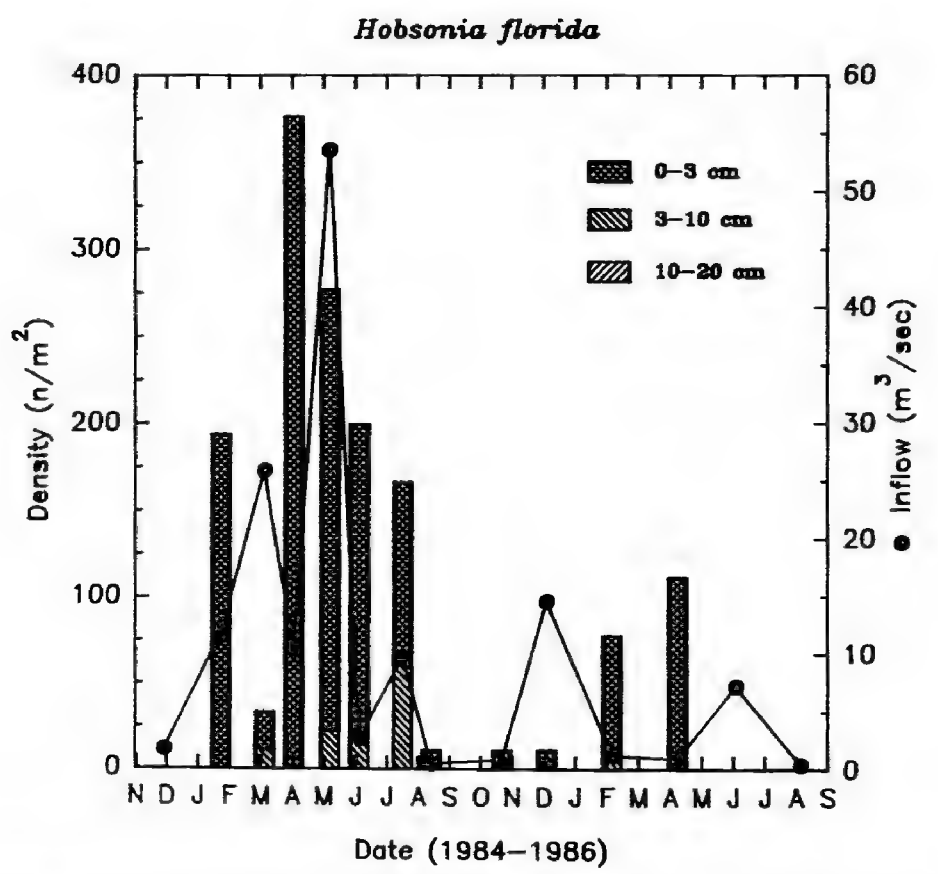


FIG. 7. *Hobsonia florida* average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

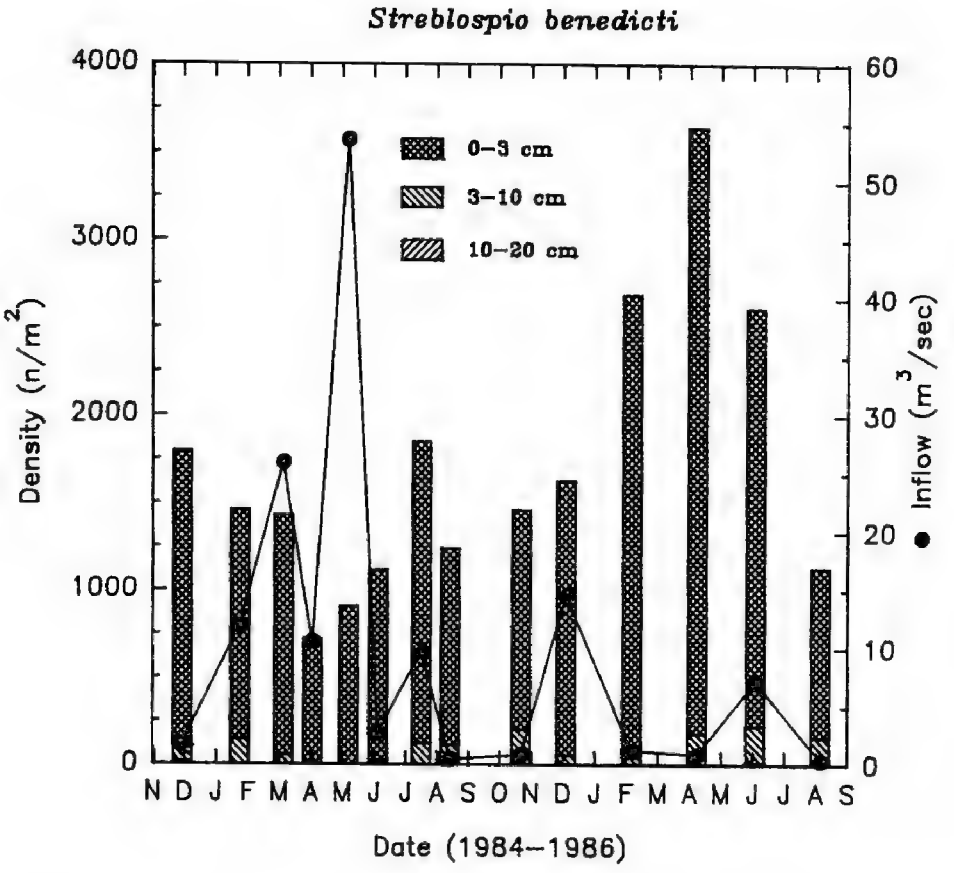


FIG. 8. *Streblospio benedicti* average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

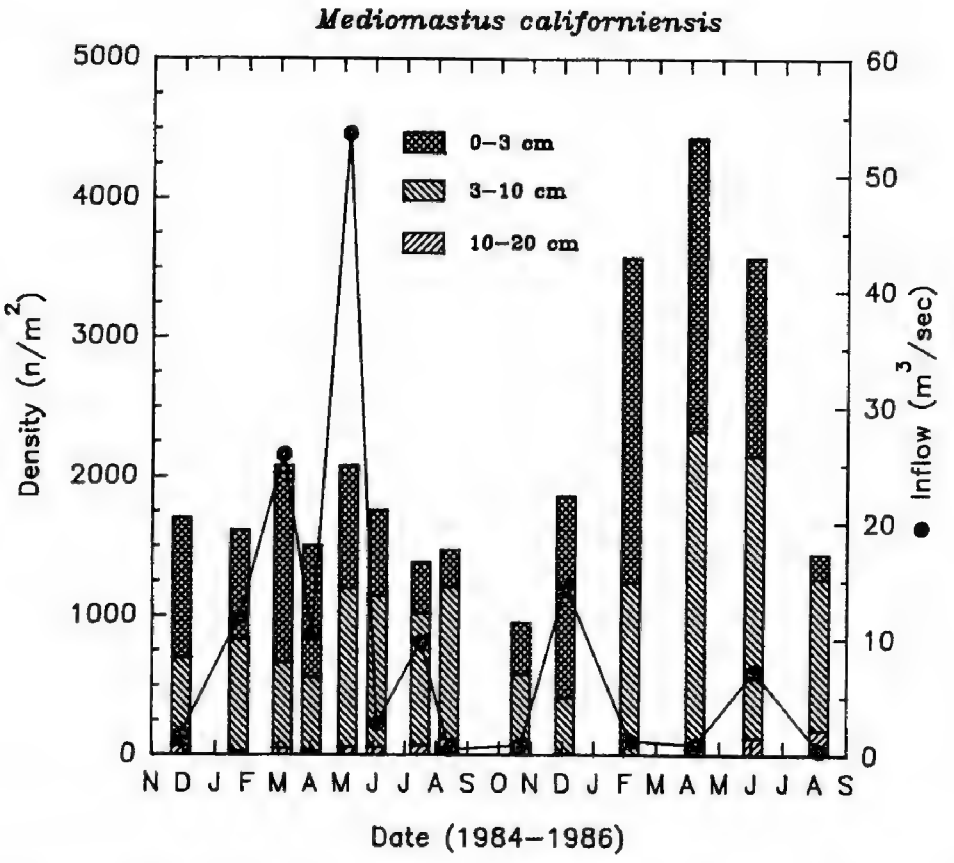


FIG. 9. *Mediomastus californiensis* average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

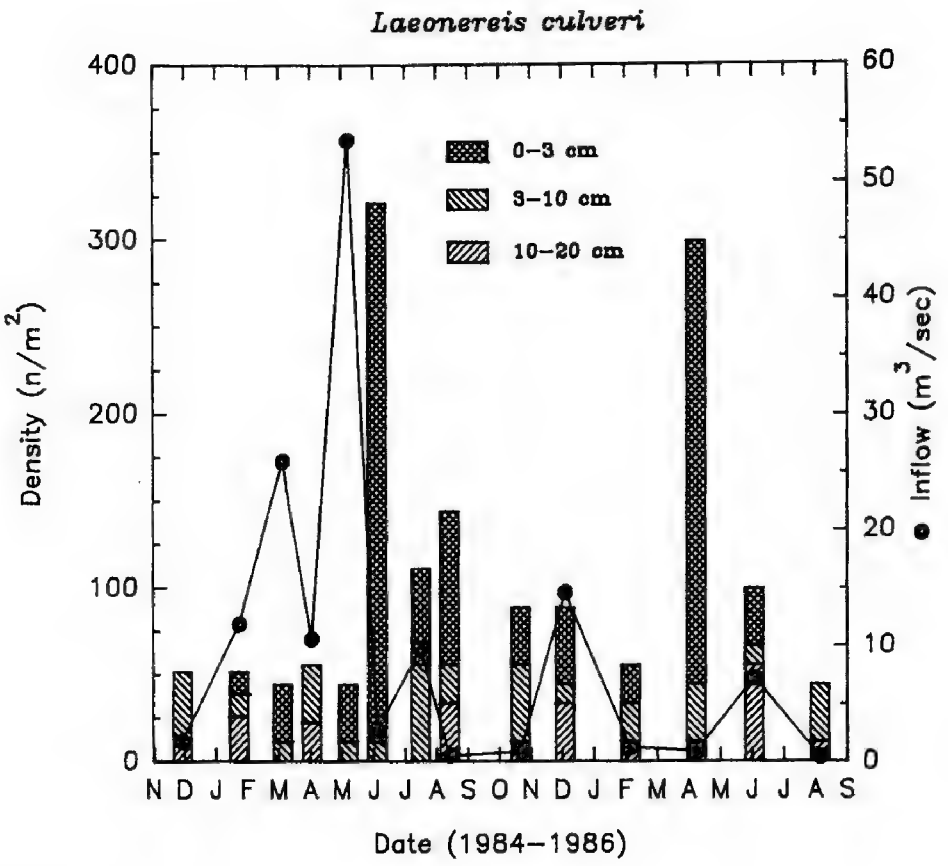


FIG. 10. *Laeonereis culveri* average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

Laeonereis culveri was not numerically dominant but it was a much larger polychaete than *S. benedicti* and *M. californiensis*. *Laeonereis culveri* occurred at all depths sampled but occasionally it was collected from only the deep core sections (Fig. 10). The density of *L. culveri* was significantly correlated with salinity ($r = 0.82$, Table 5).

The two dominant mollusks, *M. lateralis* and *M. mitchelli* accounted for most of the benthic biomass. Both species occurred throughout the study and reached their highest density in February and March 1986. Although their peak density occurred during low inflow periods, there was no significant correlation with salinity or freshwater inflow and density (Table 5). *Mulinia lateralis* is a surface dweller which was collected predominantly in the upper 0-3 cm (Fig. 11). *Macoma mitchelli* inhabited sediments from the surface to 20 cm. When *M. mitchelli* first settled in the benthos it was found mainly in the surface sediments. In December 1985, the concentration of *M. Mitchelli* at the 0-3 cm sediment was juveniles (Fig. 12). As *M. Mitchelli* matured, larger specimens were found in the deeper sections in April and June 1986.

Stations 45, 65 and 85 form a salinity gradient from low to higher salinity in the Lavaca River (Fig. 1). In each trip the salinity was lowest at station 45, increased downstream at station 65 and was highest at station 85 (Fig. 13). The salinity from November 1984 through August 1985 was lower than the period from October 1985 through August 1986. The stations also formed a

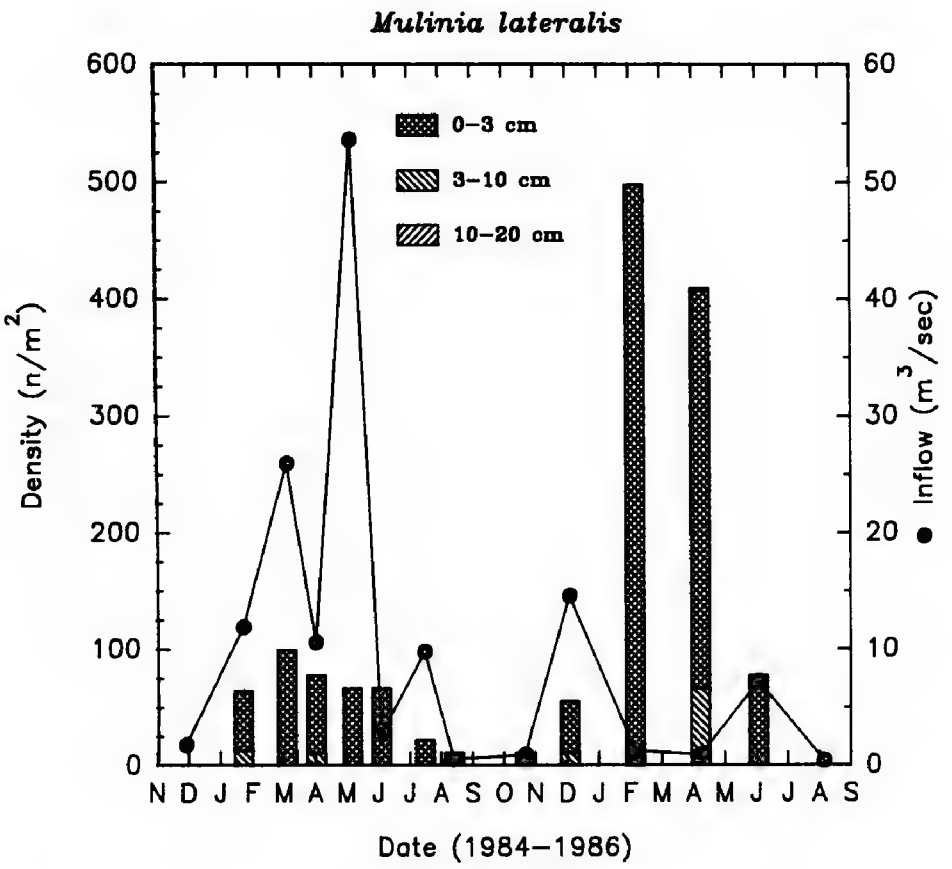


FIG. 11. *Mulinia lateralis* average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

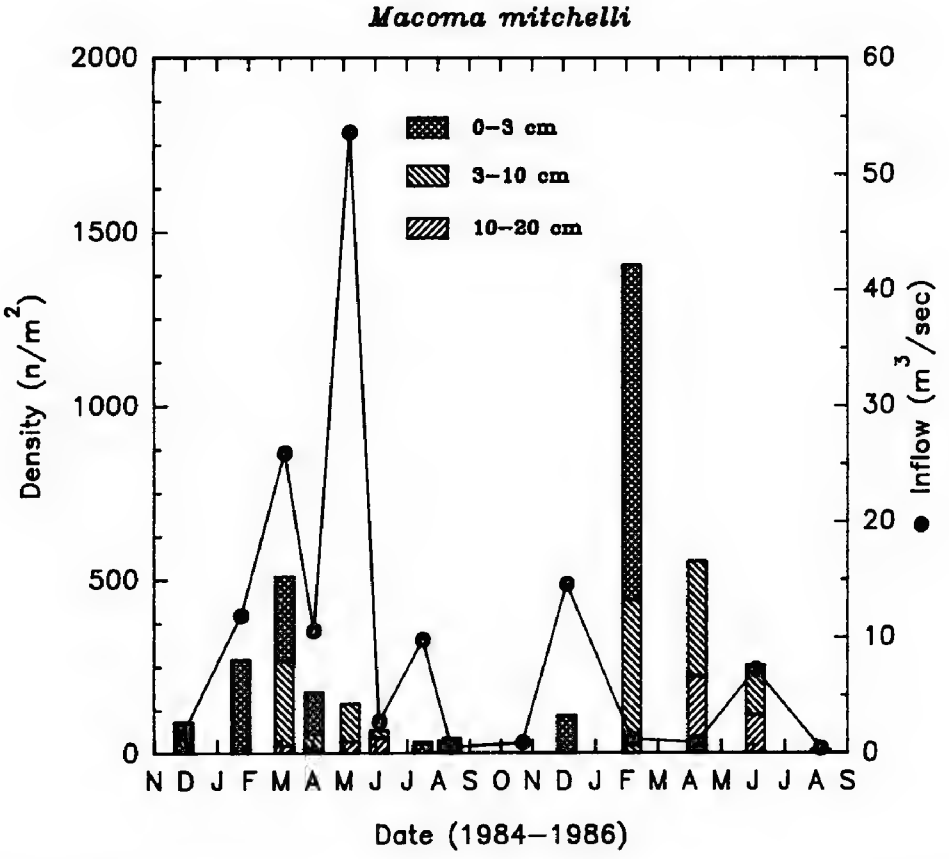


FIG. 12. *Macoma mitchelli* average abundance (n/m^2) at 0-3, 3-10, and 10-20 cm sediment depths and the 14-day average streamflow (solid line) by sampling trip from November, 1984 through August, 1986.

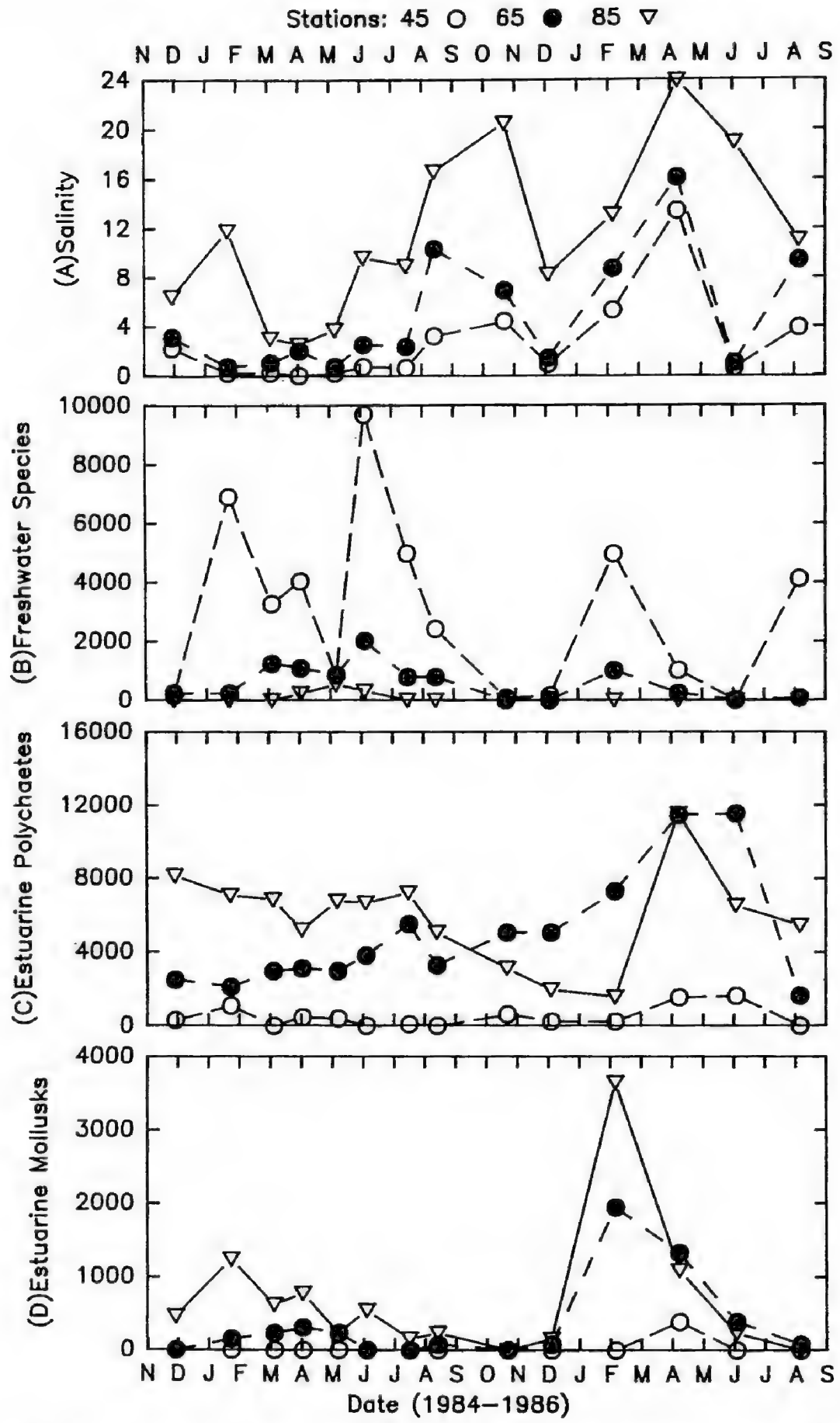


FIG. 13. Spatial and temporal response of species groups to salinity regimes. Salinity (‰) freshwater species (Chironomid larvae and *Hobsonia florida*), estuarine polychaetes (*Mediomastus californiensis, Streblospio benedicti* and *Laeonereis culveri*, and estuarine mollusks (*Mulinia lateralis* and *Macoma mitchelli*) abundance (n/m^2). Salinity gradient is from upstream to upper Lavaca Bay at stations 45, 65 and 85 for sampling trips from November, 1984 through August, 1986.

gradient of decreasing grain-size (Table 4). The dominant fauna at stations 45, 65 and 85 were freshwater and estuarine species. The freshwater or low salinity species were Chironomid larvae and *H. florida*. The estuarine species were the polychaetes: *S. benedicti*, *M. californiensis*, and *L. culveri* and the mollusks: *M. lateralis* and *M. mitchelli*. The average density of these species groups was compared with salinity changes to determine changes in areal distributions affected by fluctuation of freshwater inflow (Fig. 13). Freshwater species were abundant following the first sampling trip and remained abundant through August 1985. Because of high inflow, low salinity species were seen at station 85 in May 1985. Following August 1985, low salinity species were evident only in the river at stations 45 and 65 in February 1986 and at Station 45 in August 1986. Estuarine polychaetes and mollusks were most abundant at stations 65 and 85 and increased during Year 2. Estuarine species were present in low numbers or absent from station 45 until April and June, 1986 when the estuarine species briefly extended their range up river.

DISCUSSION

Low rainfall during the period from December, 1983 to September, 1984 caused drought conditions in the Lavaca Basin (Fig. 2). This resulted in an order from the Texas Water Commission on August 29, 1984 directing the Lavaca-Navidad River Authority to release 12×10^6 m^3 (10,000 acre-feet) of water from Lake Texana into the Lavaca Bay System (TDWR 1985). From August 31 through September 7, 1984 water was released at a rate of 21 m^3/s. Salinity decreased from 2 ‰ below Palmetto Bend Dam to 0 ‰ and from 15 ‰ near Redfish Bayou to 10 ‰. Lower Lavaca Delta and upper Lavaca Bay salinities decreased by only 1 ‰ immediately after the release (TDWR 1985). High Gulf of Mexico tides associated with tropical disturbances resulted in high tides from September 16 to 25 to push high salinity Gulf waters into the Lavaca-Tres Palacios Estuary raising salinities above those occurring prior to the release. In October 1984, a freshwater inflow event occurred which resulted in the beginning of a wet cycle which persisted throughout Year-1 of this study.

High freshwater inflow during Year-1 of this study resulted in an increase in the spatial distribution of low salinity benthic species and a decrease in estuarine and marine species in upper Lavaca Bay. The extensive distribution of chironomid larvae and *H. florida* in upper Lavaca Bay was indicative of flood conditions and increased spatial coverage of the freshwater zone.

Freshwater inflow decreased during Year-2 which reduced the freshwater influenced zone and increased the higher salinity estuarine zone. The distribution of low salinity species during Year-2 was restricted to the Lavaca River and lake stations and the stations close to Garcitas and Venado creeks. *Streblospio benedicti*, *M. californiensis*, and *L. culveri* all increased in abundance and areal distribution as a result of higher salinity during Year-2. Total

increased abundance and biomass during Year-2 is attributed to benthic community response, not only to salinity increases, but also to primary production stimulated by nutrient input into the system the preceding high inflow year. Nitrate, nitrite, and phosphate were higher in the wet year (Jones, *et al.* 1986). Chlorophyll increased in Year-1 and decreased in Year-2.

Other studies have found similar relationships between benthos and inflow. Mackin (1971) reported an increase in total benthic abundance in Menefee Lake with increased salinity. Species diversity declined from the high salinity lower bay to the low salinity upper bay and Lavaca River (Gilmore, *et al.* 1976). Lavaca Bay freshwater-influenced benthic populations increased as salinity decreased and organic carbon increased during a 30 month freshwater inflow study from January 1973 through June 1975 (Gilmore 1974; Gilmore, *et al.* 1975, 1976).

The distribution of estuarine benthos in Texas estuaries has been partitioned into zones based on species salinity preferences along a salinity gradient from freshwater inflow to marine input in several studies (Flint, *et al.* 1981; Gilmore 1974; Gilmore, *et al.* 1975, 1976; Harper 1973; Hoese 1960; Mackin 1971; Matthews, *et al.* 1974; Ladd 1951; Parker 1955, 1959; Whitledge, *et al.* 1989). The present study indicates that these arbitrary zones are not constant but fluctuate up and down the bay system depending on freshwater inflow and marine input from the Gulf of Mexico (Fig. 13).

The mollusks, *Rangia cuneata* and *Littoridina sphinctostoma*, which have been reported as abundant in the freshwater influenced zone by Gilmore, *et al.* (1976) and White, Calnan, Morton, Kimble, Littleton, McGowen and Nance (1985) were rarely found in this study. Low density, or the absence of these species, may be attributed to patchy distribution or a slow response of these species to environmental fluctuation. For example, the year prior to this study was a drought year which may have limited the range of *R. cuneata* and *L. sphinctostoma* to areas upstream which were not sampled. In other Lavaca Bay studies chironomid larvae and/or *H. florida* were reported to be associated with low salinities (Blanton, Culpepper, Bischoff, Smith and Blanton 1971; Gilmore, *et al.* 1976; Mackin 1971). During Mackin's study the most upriver site in the upper Menefee Lake remained influenced by freshwater for the entire study while the lower Menefee Lake, Redfish Lake and lower river stations changed from a low salinity zone to a moderate salinity zone. The fauna changed from a freshwater community to a marine community at about the same rate the salinity increased (Mackin 1971).

Vertical sectioning of the core samples at 0-3, 3-10, and 10-20 cm enabled us to characterize benthic species by their preference for surface or sub-surface sediments. The numerically dominant surface dwelling species collected during this study were *S. benedicti, H. florida*, chironomid larvae, and *M. lateralis. Mediomastus californiensis, L. culveri* and *M. mitchelli* were occasionally abundant at the surface but had a preference for the deeper sediment. The mean percent abundance and biomass by depth indicated that most of the macrofauna abundance were found in the surface sediments and the greatest

biomass occurred sub-surface at the 10-20 cm depth. Flint and Kalke (1986) found similar vertical distribution patterns for benthic macrofauna in Corpus Christi Bay. High densities of small organisms with a short turnover rate in the surface sediments in the long term make more biomass available to higher trophic levels, *i.e.*, finfish and shellfish. The deeper dwelling animals are less abundant larger animals which are not easily incorporated into the food chain because of their deep burrowing activities.

Although the biomass and abundance of the dominant mollusk species did not significantly correlate with inflow, they did increase with salinity (Table 5). Density and biomass of all mollusks increased significantly with increasing salinity (Table 6). Other environmental factors are also important. Total biomass increased with increasing temperature. Whereas, molluskan biomass increased with salinity, other macrofauna increased with increasing sand content and chlorophyll concentration in the overlying water (Table 6). Chlorophyll concentration increased with increasing inflow and decreased with increasing salinity. Considering the large increase of filter feeding mollusks with increasing salinity, the decrease in chlorophyll concentration could be due to increased consumption by the benthic community.

The variability in freshwater inflow results in predictable changes in the estuary. Flood conditions lower salinity and introduce nutrient rich waters into the estuary. This happens rapidly within 14 days. Within one month, the spatial extent of the low salinity fauna is increased. The freshwater fauna may even replace the estuarine fauna. High nutrient and chlorophyll levels correlate with high benthic productivity of predominantly low salinity and estuarine species. This community can deplete the surface water phytoplankton bloom since it is dominated by filter feeders. This can be followed by a transition to drought conditions and low inflow regimes resulting in higher salinities, lower nutrients, and higher densities of estuarine fauna. The results of benthic sampling programs depend on the inflow conditions existing prior and during the study. This indicates long-term studies are necessary to understand the relationship between freshwater inflow and benthic community structure and productivity.

ACKNOWLEDGMENTS

We would like to thank Lynn Tinnin, Julie Findley, and Wen Lee for their assistance in field sampling and laboratory data processing. Sampling assistance was given by Hayden Abel, Noe Cantu, Don Gibson, John Turany and Billy Slingerland of the boat crew. William Longley of the TWDB supplied us with freshwater inflow data. The work was funded through the Water Research Planning Fund, and administered by the TWDB under interagency contract no. IAC (86-87) 0757, TWDB contract 55-61011. University of Texas at Austin Marine Science Institute Contribution No. 823.

LITERATURE CITED

BLANTON, W.G., T.J., CULPEPPER, H.W. BISCHOFF, A.L. SMITH and C.J. BLANTON. 1971. A study of the total ecology of a secondary bay (Lavaca Bay). Report to the Aluminum Co. of America. Aluminum Co. of America, Point Comfort, Texas. 306 pp.

BUCKNER, H.D., E.R. CARILLO and H.J. DAVIDSON. 1987. Water resources data, Texas, water year 1987, Colorado River Basin, Lavaca River Basin, Guadalupe River Basin, Nueces River Basin, Rio Grande River Basin, and intervening coastal basins. Report to the U.S. Geological Survey Water Resources Division, TX-87-3. United States Geological Survey, Austin, Texas. 424 pp.

FLINT, R.W. and R.D. KALKE. 1986. Biological enhancement of estuarine benthic community structure. *Marine Ecology — Progress Series*. 31:23-33.

FLINT, R.W., R.D. KALKE, and S.C. RABALAIS. 1981. Quantification of extensive freshwater input to estuarine benthos. Report to the Texas Water Development Board. The University of Texas Marine Science Institute, Port Aransas, Texas. 55 pp.

FOLK, R.L. 1964. *Petrology of Sedimentary Rocks*. Hemphill's, Austin, Texas. 154 pp.

GALTSOFF, P.S. 1931. A survey of oyster bottoms in Texas. Bureau of Fisheries Investigational Report No. 6. Washington, DC. 30 pp.

GILMORE, G. 1974. An ecological study of benthic macro-invertebrates on Lavaca Bay, Texas. Report to the Texas Water Development Board. Texas Parks and Wildlife Department, Austin Texas. 26 pp.

GILMORE, G., G. MATTHEWS and G. CLEMENTS. 1975. An ecological study of the plankton and benthos in Lavaca Bay, Texas. Report to the Texas Water Development Board, September 1973 July 1974. Texas Parks and Wildlife Department, Austin Texas. 58 pp.

GILMORE, G., J. DAILEY, M. GARCIA, N. HANNEBAUM, and J. MEANS. 1976. A study of the effects of fresh water on plankton, benthos, and nekton assemblages of the Lavaca Bay system, Texas. Report to the Texas Water Development Board, January 1973-1975. Texas Parks and Wildlife Department, Austin Texas. 113 pp.

HARPER, D.E., Jr. 1973. The distribution of benthic and nektonic organisms in undredged control areas of San Antonio Bay, Texas. Report to the U.S. Army Engineer District. Texas A&M Research Foundation. College Station, Texas. 157 pp.

HOESE, H.D. 1960. Biotic changes in a bay associated with the end of a drought. *Limnology and Oceanography*. 5:326-336.

HOLLAND, J.S., N.J. MACIOLEK, and C.H. OPPENHEIMER. 1973. Galveston Bay benthic community structure as an indicator of water quality. *Contributions in Marine Science*. 17:169-188.

———, ———, R.D. KALKE and C.H. OPPENHEIMER. 1974. A benthos and plankton study of the Corpus Christi, Copano and Aransas Bay systems II. Report on data collected during the period July, 1973 and April, 1974. Report to the Texas Water Development Board. The University of Texas Marine Science Institute, Port Aransas Texas. 121 pp.

———, ———, ———, L. MULLINS and C.H. OPPENHEIMER. 1975. A benthos and plankton study of the Corpus Christi, Copano and Aransas Bay systems and summary of the three-year project. Report to the Texas Water Development Board. III. Report on data collected 7/74-5/75. The University of Texas Marine Science Institute, Port Aransas, Texas. 174 pp.

JONES, R.S., J.J. CULLEN, R.G. LANE, W. YOON, R.A. ROSSON, R.D. KALKE, S.A. HOLT and C.R. ARNOLD. 1986. Studies of freshwater inflow effects on the Lavaca River Delta and Lavaca Bay, Texas. Report to the Texas Water Development Board. The University of Texas Marine Science Institute, Port Aransas, Texas. 423 pp.

LADD, H.S. 1951. Brackish water and marine assemblages of the Texas coast, with special reference to mollusks. *Publications of the Institute of Marine Science, The University of Texas*. 2:125-164.

MACKIN, J.G. 1971. A study of the effect of oilfield brine effluents on biotic communities in Texas estuaries. Report to the Humble Oil and Refining Co. and Amoco Production Co. Texaco Inc., Texas A&M Research Foundation, College Station, Texas. 73 pp.

MATTHEWS, G.A., C.A. MARCIN and G.L. CLEMENTS. 1974. A plankton and benthos survey of the San Antonio Bay system March 1972-July 1974. Report to the Texas Water Development Board. Texas Parks and Wildlife Department, Austin, Texas. 75 pp.

MOORE, H.F. 1907. Survey of oyster bottoms in Matagorda Bay, Texas. United States Bureau of Fisheries Document No. 610. 86 pp.

MOORE, H.F. and E. DANGLADE. 1915. Condition and extent of the natural oyster beds and barren bottoms of Lavaca Bay, Texas. Report of the U.S. Commissioner of Fisheries for 1914. Hugh H. Smith Commissioner. 45 pp.

MUNRO, G.J. 1961. General ecological survey of the Matagorda area, September 1, 1960 — December 31, 1961. Report for Project No. M-4-R-3, Job D-2, Texas Parks and Wildlife Department, Austin, Texas. 6 pp.

PARKER, R.H. 1955. Changes in the invertebrate fauna, apparently attributable to salinity changes in the bays of central Texas. *Journal of Paleontology*. 29:193-211.

PARKER, R.H. 1959. Macro-invertebrate assemblages of central Texas coastal bays and Laguna Madre. *Bulletin of the American Association of Petroleum Geologists*. 43:2100-2166.

SHEPARD, F.P. 1953. Sedimentation rates in Texas estuaries and lagoons. *Bulletin of the American Association of Petroleum Geologists*. 37:1919-1934.

TEXAS DEPARTMENT OF WATER RESOURCES (TDWR). 1980. Lavaca-Tres Palacios Estuary: a study of the influence of freshwater inflows. TDWR, Austin, Texas. LP-106.

TEXAS DEPARTMENT OF WATER RESOURCES (TDWR). 1982. The influence of freshwater inflows upon the major bays and estuaries of the Texas Gulf Coast: Executive Summary. TDWR, Austin, Texas. 53 pp. Appendices I-VI.

TEXAS DEPARTMENT OF WATER RESOURCES (TDWR). 1985. Investigation of the effects of releases of water from Lake Texana, Lavaca River Basin, on August 31 September 7, 1984. TDWR, Austin, Texas. 25 pp. Appendices A-D.

U.S. GEOLOGICAL SURVEY (USGS). 1980. Water resources data for Texas Colorado River Basin, Lavaca River Basin, Guadalupe River Basin, Nueces River Basin, Rio Grande Basin, and intervening coastal basins. USGS Water Data Report TX-80-3. USGS, Austin, Texas. 298 pp.

WHITE, W.A., T.R. CALNAN, R.A. MORTON, R.S. KIMBLE, T.G. LITTLETON, J.H. McGOWEN and H.S. NANCE. 1985. Submerged lands of Texas, Port Lavaca area: sediments, geochemistry, benthic macroinvertebrates and associated wetlands. Report by Bureau of Economic Geology, Austin, Texas. 309 pp.

WHITLEDGE, T.E., A. AMOS, R. BENNER, E. BUSKEY, K. DUNTON, S. HOLT, R. KALKE, P. MONTAGNA, P. PARKER, D. STOCKWELL and W.B. YOON. 1989. Final Report Nitrogen Process Studies (NIPS): The effect of freshwater inflow on benthos communities and dynamics. Report to the Texas Water Development Board. The University of Texas Marine Science Institute, Port Aransas, Texas. 370 pp.

Contributions in Marine Science. (1991) Vol. 32.

A LAMINAR-FLOW CHOICE CHAMBER FOR TESTING THE RESPONSES OF POSTLARVAL PENAEIDS TO OLFACTANTS

Mark C. Benfield[1] and David V. Aldrich[1]

Department of Wildlife and Fisheries Sciences
Texas A&M University
College Station, Texas 77843

ABSTRACT

A laminar-flow dichotomous choice chamber has been developed for testing the responses of postlarval brown shrimp, *Penaeus aztecus*, and white shrimp, *Penaeus setiferus*, to olfactants. Test animals were provided with equal exposure to two discrete olfactant streams which were separated by a steep gradient in a laminar current. Studies with fluorescein dye indicated that the separation of olfactant streams was stable, reproducible and unaffected by the passage of a shrimp through the chamber. Shrimp responses to odor appears to be unbiased by the chamber geometry. Our design overcomes turbulent mixing and concentration fluctuation problems inherent in most Y-maze designs. The initial positioning of the test animal along the boundary of the two water streams ensures equal exposure to both olfactants. This factor coupled with a relatively long distance in which to experience both odors provides each animal with sufficient information to arrive at an unbiased choice. Postlarval white shrimp selected the dye with a significantly greater frequency than the control, while brown shrimp did not respond to the dye stimulus. Brown shrimp demonstrated a significant preference for estuarine water over synthetic seawater of the same salinity. This chamber is currently being used to identify the sources of components of estuarine water which are attractive to these two species.

INTRODUCTION

The mechanisms by which postlarval penaeid shrimp locate their estuarine nursery grounds are poorly understood (Mair, Watkins and Williamson 1982). Postlarvae are thought to use horizontal salinity gradients to orient towards estuaries (Keiser and Aldrich 1976; Mair 1980) while subsequent movement into these systems may be accomplished by selective tidal transport possibly synchronized by salinity differentials between tidal currents (Hughes 1969, 1972), or hydrostatic pressure (Forbes and Benfield 1986). Documentation of successful recruitment into hypersaline estuaries (Penn 1975; Mair, *et al.* 1982) suggests that other cues may be important. Once shrimp gain access to

---

[1] Present Address: Department of Marine Biology, Texas A&M University at Galveston, P.O. Box 1675, Galveston, Texas 77553-1675.

estuarine systems, the mechanisms by which they locate nursery habitats (salt marshes, seagrass beds and mangroves) are unknown.

The importance of chemical odors to diadromous salmon during their spawning migrations has been the subject of intense study (Hasler, Scholz and Horral 1978; Johnsen and Hasler 1980; Stabell 1984); however, the role played by odors in the orientation of postlarval penaeids remains conjectural and has not received serious experimental attention. Convincing evidence of the importance of odor cues to migratory species such as American eels, *Anguilla rostrata,* has been presented by Sorensen (1986). Autochthonous and allochthonous chemicals in estuarine waters could provide cues which, in addition to salinity, might assist postlarvae in locating and entering estuaries and nursery habitats within these systems. Odum (1970) suggested that estuarine organisms may use road maps of dissolved organic material to orient towards estuaries. Earlier work by Kristensen (1964) determined that *Penaeus aztecus* postlarvae were preferentially attracted to bay water over seawater and that the bay water lost its attractiveness following passage through a charcoal filter. Ruello (1973) suggested that young school prawns, *Metapenaeus macleyi*, utilized dissolved organic compounds in estuarine water to direct their movements into the estuary. The most convincing case for a response to odor was presented by Mair (1980) and Mair, *et al.* (1982) who found that postlarval *P. californiensis* and *P. brevirostris* were attracted to estuarine water over artificial seawater of the same salinity in a gradient choice chamber.

Estuarine systems frequently receive high pollutant loadings from surrounding industrial and agricultural activities. Avoidance of pollutants has been documented for many aquatic species including salmon *Oncorhynchus kisutch* (Maynard and Weber 1981), the European brown shrimp *Crangon crangon* (Moelenberg and Kioeboe 1973), and grass shrimp *Palaemonetes pugio* (Hansen, Schimmel and Keltner 1973).

As part of an investigation into the potential role of odor cues in the estuarine recruitment of postlarval brown shrimp *Penaeus aztecus* and white shrimp *P. Setiferus* into Galveston Bay, we required a choice chamber for measuring the unconditioned responses of these species to olfactants. A general review of chamber designs may be found in Scarfe, Steel and Rieke (1985). The gradient tanks developed by Keiser and Aldrich (1973) and used to examine postlarval responses to salinity (Keiser and Aldrich 1976; Mair 1980; Mair, *et al.* 1982) were unsuitable for olfactant choice studies because of the confounding effect of salinity (density) necessary to maintain stability. Most conventional chamber designs including Y-maze systems provide little control over dispersion and concentrations of olfactants in the common section of the chamber (Scarfe, *et al.* 1985). In addition, potential bias towards one olfactant caused by unequal pre-release exposure of the test animal has not been addressed. While the octagonal fluvarium developed by Scarfe, *et al.* (1985) provided greater control over concentration and permitted testing of multiple olfactants, the olfactants flowed perpendicularly to the normal axis

of animal movement and analysis of results was complicated by the statistical interdependence of sample observations. The objective of this study was to develop and evaluate a chamber suitable for measuring the attraction and avoidance of postlarval shrimp to estuarine olfactants under laboratory conditions.

The present paper describes the construction and operation of a single animal choice chamber which provides exposure to two adjacent olfactants separated by a sharp boundary without physical partitioning. The chamber design provides test animals with information about the nature and orientation of the olfactants prior to release. Dye flow studies indicate that olfactant concentrations are predictable, reproducible and stable along the length of the chamber. Experiments with and without a dye (bromthymol blue) suggest that the chamber provides unbiased quantitative estimates of postlarval shrimp attraction to a stimulus and appears to be suitable for testing their responses to natural and pollutant olfactants. Tests with natural estuarine water confirmed both the attractiveness of this odor source to postlarval brown shrimp and the utility of this device for assessing the olfactory preferences of postlarval penaeid shrimps.

MATERIALS AND METHODS

Chamber Construction and Operation

The apparatus consists of a seawater and olfactant delivery system (Fig. 1) and a laminar flow choice chamber (Fig. 2). The chamber is constructed of 5.6 mm plexiglass sheet attached to an 11.8 mm plexiglass base with a solution of plexiglass in methylethyl ketone. All plexiglass joints are sealed with silicone along their external surfaces while other chamber fixtures such as Nytex™ screens are attached with cyanoacrylate cement.

Carrier water consisting of synthetic seawater (Fritz Supersalt™ dissolved in 18 MΩ deionized water and passed through 5μ and activated carbon filters) circulates between a 150 liter reservoir and a 20 liter head tank (Fig. 1). From the head tank, the carrier water is gravity fed *via* a metered Tygon line equipped with a flow-controller and enters the chamber at the center of its upstream end (Fig. 1). Once in the chamber, carrier water passes through a compartment containing 3 mm diameter glass beads which reduce channelling and ensure an equal distribution of water to either side of the central partition (Fig. 2). The water is divided into two parallel streams by a central partition located immediately downstream of the glass beads and each stream flows through a series of compartments which are separated by Nytex mesh (Fig. 2). Olfactants are pumped from reservoirs through 7.32 m heat exchangers located in the head tank and are introduced into opposite sides of the partition in the second compartments of the chamber (Figs. 1, 2). The coiled glass heat exchangers eliminate temperature differentials induced by passage of the olfactants through the peristaltic pumps. After leaving these heat exchangers the olfactant solutions did not differ by more than 0.2° C and were within 0.1° C of the carrier water temperature. During the development phase of this study, gradual heating of the peristaltic pumps resulted in density-driven mixing of the two olfactant streams when the temperature differential exceeded 0.2° C. Glass air injectors located in the next two compartments facilitate turbulent mixing of the carrier water and olfactants (Fig 2). Passage of the carrier water/olfactant mixtures through subsequent Nytex screens eliminates turbulence and promotes laminar flow. When the parallel water streams emerge from the bevelled end of the central partition, laminar flow maintains their olfactory integrity. This lack of mixing results in a sharp boundary between the two streams despite the absence of any physical partition. An animal holding device

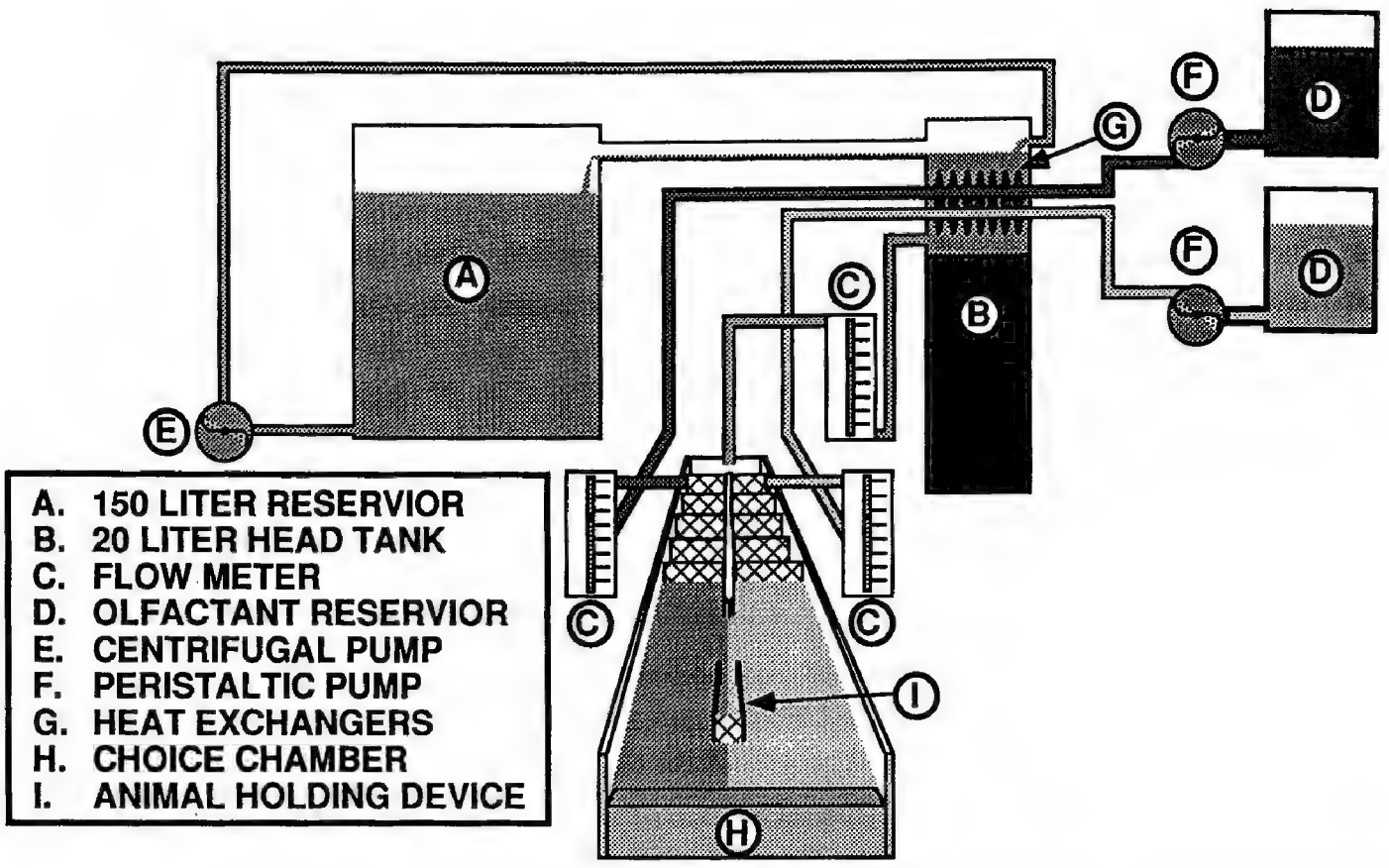


FIG. 1. Choice chamber and associated water and olfactant delivery system. Air injection system and vertical gate of animal holding device omitted for clarity.

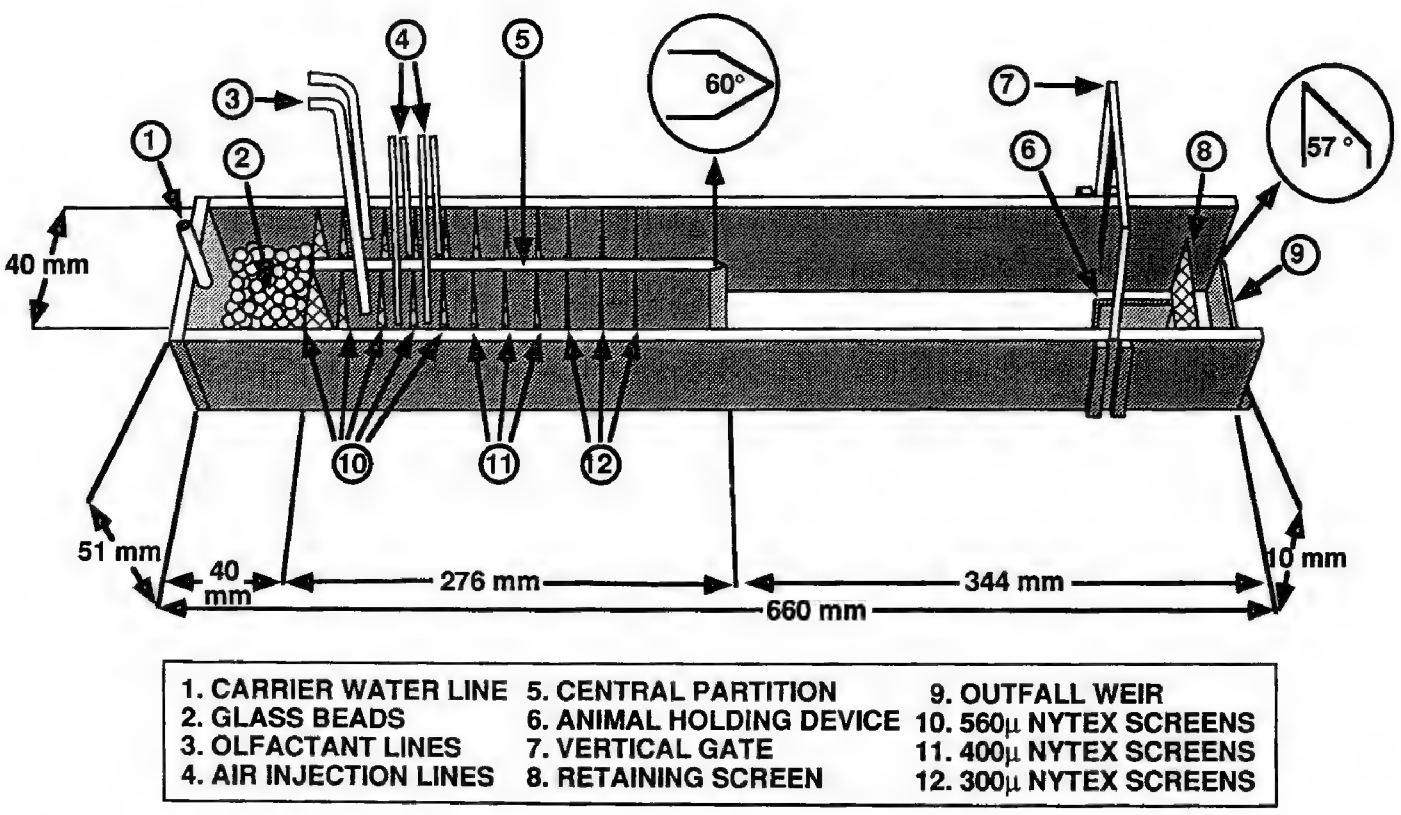


FIG. 2. Schematic diagram of the plexiglass choice chamber. Exploded circles indicate the angles of the central partition and downstream weir.

constructed of 0.2 mm plastic sheet and equipped with a vertical sliding gate on its upstream end is precisely positioned to intercept the boundary of the two water streams along its long axis (Fig. 2). This allows simultaneous equal exposure of the animal to both olfactants at the onset of each trial. Downstream of the animal holding device water leaves the chamber after passage over a bevelled outfall weir.

A future requirement for testing organic pollutants led to the construction of a duplicate chamber made primarily of glass. This chamber had similar dimensions to the original plexiglass chamber and incorporated a longer (50 mm long) common section at the upstream end which provided superior distribution of carrier water to either side of the central partition and eliminated the requirement for glass beads. Ten 400 μ Nytex mesh screens were placed on either side of the central partition. The central partition was constructed of 2.7 mm thick plexiglass with a 32° bevel on both ends. Flow rates of carrier and test water were the same as in the plexiglass chamber.

Dye Flow Studies

In order to establish a standard testing protocol we required information about the degree of mixing downstream from the partition; the presence of possible chamber bias; the time required to establish a stable pair of water streams following reversal of the chamber side receiving the olfactant; and the effect of animal movement on the stability of the odor streams. These data were obtained with fluorescein disodium salt, a dye which simulated the passage of an olfactant through the chamber.

The degree of mixing between the two water streams was measured by introducing carrier water into the chamber at 97.4 ml•min^{-1} while 10.8 mg•L^{-1} fluorescein (disodium salt) dissolved in 30 ‰ carrier water, and carrier water alone, were pumped into opposite sides of the chamber at flow rates of 19.5 ml•min^{-1} (fluorescein) and 19.4 ml•min^{-1} (carrier water). The total chamber flow rate of 136.3 ml•min^{-1} divided by the cross sectional area resulted in a theoretical current velocity of 5.7 mm•s^{-1}. A series of ten paired water samples were withdrawn from the middle of each stream at 20 mm intervals along a 180 mm section beginning 10 mm above the upstream edge of the animal holding device and extending beyond the downstream edge of the central partition. The left and right samples forming each pair were collected simultaneously. After each group of samples had been collected the side of the chamber receiving the dye was reversed and the flow was allowed to stabilize for 10 minutes. Eight groups of samples were collected; four for each dye-side combination. Sample absorbance at 500 nm was used to estimate the fluorescein concentration.

Estimates of the time required for each stream to stabilize following a reversal of the dye feed line were obtained by withdrawing simultaneous paired samples at 1 min intervals over a period of 8 min, from the middle of each stream at a point 10 mm upstream of the holding device, following reversal of the olfactant lines.

The impact of animal movement on the stability of the dye stream was assessed visually during an experiment to estimate chamber bias in postlarval white shrimp, *Penaeus setiferus* and brown shrimp, *P. aztecus*. The use of dye (bromthymol blue) also provided an opportunity to assess attraction and avoidance to a synthetic substance. Bromthymol blue was used instead of fluorescein because the postlarvae were more readily visible in the pale blue dye stream. All shrimp were captured from the surf zone of Galveston Island and held in filtered aquaria containing 30 ‰ carrier water. Each species was tested with 55-100 trials, depending upon postlarval availability. During each trial bromthymol blue (0.002% in 30 ‰ carrier water) was pumped into the randomly determined left or right side of the chamber at 20 ml•min^{-1} and the dye stream was allowed to stabilize for 5 min. Carrier water (30 ‰) was delivered into the end of the chamber at approximately 92 ml•min^{-1} and into the side opposite the dye at 20 ml•min^{-1} in order to provide a current velocity of 5.5 mm•s^{-1}. One postlarval shrimp was introduced to the holding chamber with the gate down (closed) when the dye stream had stabilized. After a 60 s habituation period the gate was raised and the animal was permitted to make a choice between

the two water streams. We arbitrarily decided that a choice had been made when the animal crossed a line 20 mm upstream from the bevelled edge of the central partition. During each trial the stability of the dye-carrier water interface was noted. Observations included the time required to make a choice after leaving the holding chamber, the course taken including the frequency of turning and the nature of the final choice. Each postlarva was then captured and preserved in 10% formalin for subsequent measurement and taxonomic confirmation. The outcomes of replicate trials were evaluated with the χ^2 statistic to test three null hypotheses: independence of attraction to dye-carrier water and attraction to the left/right chamber side; equal attraction to dye and carrier water; and equal attraction to the left and right sides of the chamber.

Estuarine Water Studies

Estuarine water was collected from the Sportsman's Road salt marsh located in West Galveston Bay, Texas. Samples were collected during or shortly after an ebb tide near a conduit which channels water from the marsh into West Bay. Approximately 25 L of water were transported in a glass carboy to the laboratory where it was filtered through a 1 μ filter and adjusted to 30 ‰ with synthetic salt or distilled water. Three replicate experiments consisting of 44-50 individual shrimp trials were conducted using the glass chamber described previously. Estuarine water was assigned to the left and right sides of the chamber with equal frequency; however, the assignment sequence was randomly determined. Experimental protocols followed those described for bromthymol blue with modifications. At the end of each trial the olfactant supply was shut off while the shrimp was removed and the chamber was rapidly drained and flushed with 400 ml of carrier water. The latter procedure served to flush and refill the system. The olfactant supply was restarted and the two streams were permitted to stabilize for 2 minutes at which time a new test animal was introduced to the holding device for a 1 min habituation period. These studies were conducted at 25 ‰ salinity using postlarval *P. aztecus* which had been collected from a Galveston Island beach and which had been acclimated to the test salinity and held for 48-96 h under constant illumination. Experiments took place between May 1-10, 1990.

RESULTS

Chamber Performance

Dye marker testing indicated that the chamber maintained effective separation of water types downstream from the central partition (Fig. 3). The concentration of the stock fluorescein solution and its input rate relative to the carrier water provided an empirical concentration estimate of 3.1 $mg \cdot L^{-1}$. Overlap of the 95% confidence intervals indicates that the measured concentrations did not differ significantly from the empirical estimate (Fig. 3). Similarly, the dye concentrations in the carrier water streams were not significantly different from 0 $mg \cdot L^{-1}$ above the 30 mm sampling point with the exception of the 70 mm samples when the dye was on the left side and in that instance, the difference was less than 0.1 $mg \cdot L^{-1}$ (Fig. 3). Concentration patterns in the dye streams were similar regardless of which side received the dye (Fig. 3).

Comparisons of the dye concentrations on opposite sides of the chamber revealed a low degree of mixing in the middle and upper sections with a slight increase evident in the vicinity of the holding device (Fig. 3). Despite this

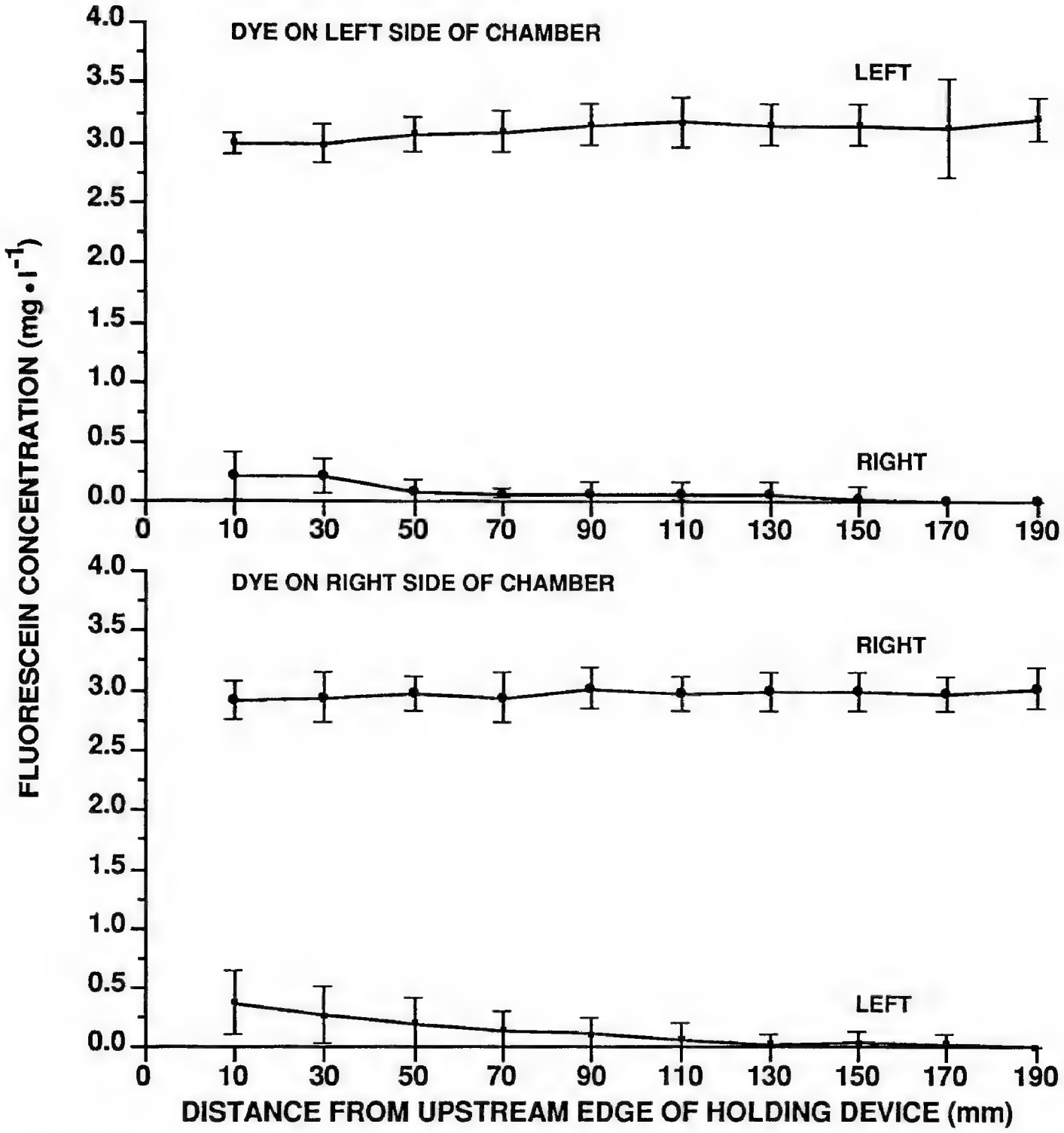


FIG. 3. Mean fluorescein concentrations on the left and right sides of the chamber when the dye is on the left side (upper plot) and right side (lower plot). Vertical bars are 95% confidence intervals on the mean.

mixing, the mean concentrations in the dye and carrier streams at the 10 mm sampling point remained significantly different (t-test, $p < 0.001$).

Between 2-5 min was required for the water streams to reestablish following reversal of the dye input side (Fig. 4). The largest time for reestablishment occurred in left to right shifts when the mean dye concentration on the right side did not differ significantly from the initial left side mean until 5 min had elapsed ($p = 0.182$).

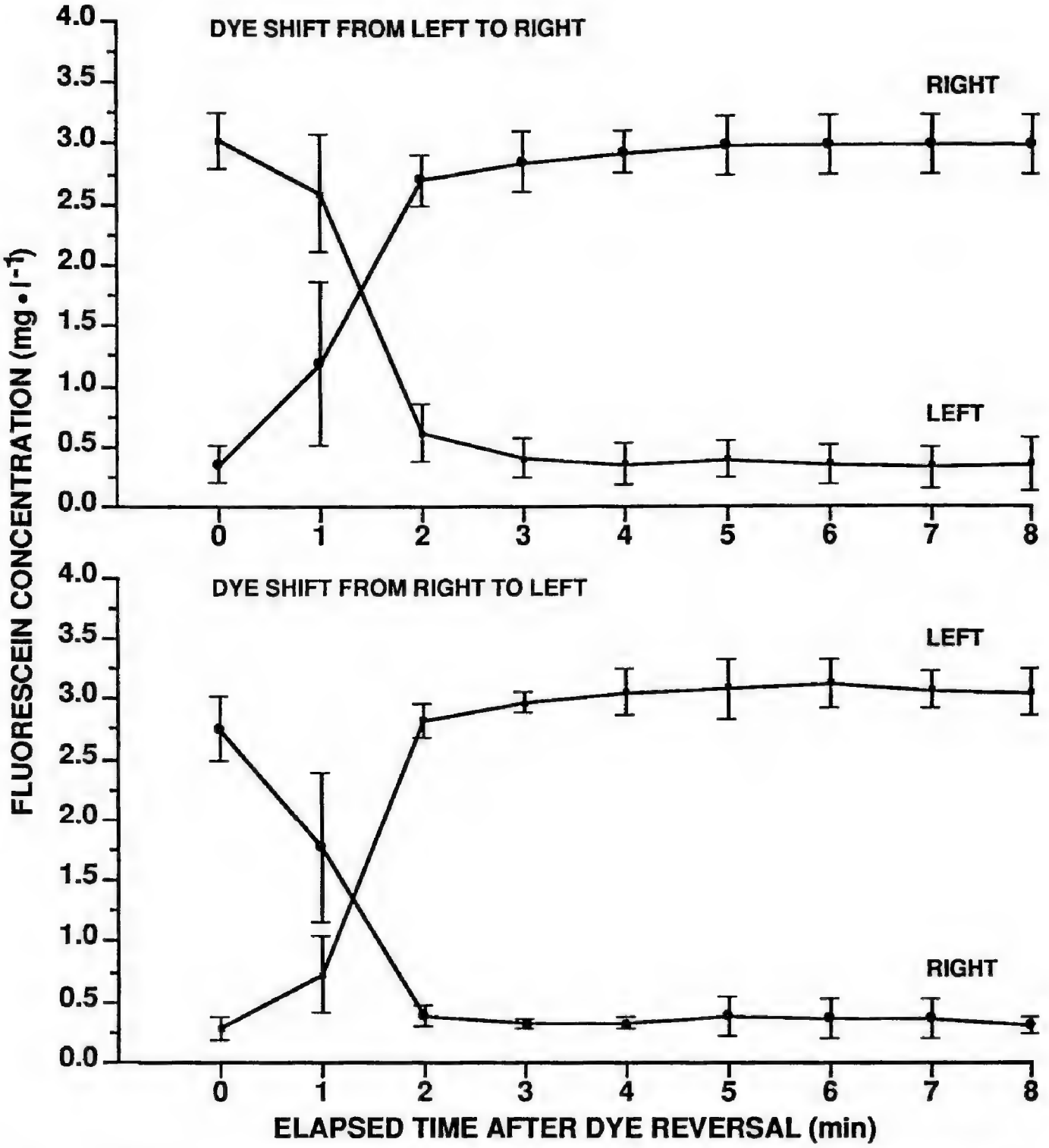


FIG. 4. Mean midstream fluorescein concentrations on the left and right sides of the chamber at one minute intervals following reversal of the dye input side. Samples collected 10 mm upstream of the animal holding device. Vertical bars indicate 95% confidence intervals on the mean.

Postlarval Behavior

The chamber appeared to be free of factors which might have contributed to shrimp bias toward either side. In two experiments using *P. aztecus* in the absence of any olfactant, there was no significant difference in the numbers of postlarvae which selected the left or right sides of the chamber (Table 1). In subsequent experiments with bromthymol blue, neither species exhibited significant chamber side preference.

TABLE 1

Summary of the responses of *P. aztecus* to chamber side and two treatments (bromthymol blue and estuarine water) and of *P. setiferus* to bromthymol blue in a laminar-flow choice chamber.

Species[1]	Treatment	n	Responses: Chamber Side[2]		Responses: Treatment[3]		Statistics: Independence	Statistics: Chamber Bias	Statistics: Treatment
P. aztecus 11.1±0.2 mm	None	90	R 43				χ^2 = 0.178 p = 0.673	χ^2 = 0.178 p = 0.673	-
			L 47						
P. aztecus 11.3±0.1 mm	None	60	R 31				χ^2 = 0.067 p = 0.796	χ^2 = 0.067 p = 0.796	-
			L 29						
P. setiferus 7.8±0.1 mm	Bromthymol Blue	100	R 50	C	29		χ^2 = 0.000 p = 1.000	χ^2 = 0.000 p = 1.000	χ^2 =17.640 p = 0.0001
			L 50	T	71				
P. aztecus 10.6±0.2	Bromthymol Blue	57	R 26	C	24		χ^2 = 0.439 p = 0.508	χ^2 = 0.439 p = 0.508	χ^2 = 1.421 p = 0.233
			L 31	T	33				
P. aztecus 10.3±0.2 mm	Bromthymol Blue	55	R 30	C	28		χ^2 = 0.455 p = 0.500	χ^2 = 0.455 p = 0.500	χ^2 = 0.018 p = 0.893
			L 25	T	27				
P. aztecus 12.0±0.3	Estuarine Water	50	R 21	C	16		χ^2 = 0.196 p = 0.658	χ^2 = 1.280 p = 0.258	χ^2 = 6.480 p = 0.011
			L 29	T	34				
P. aztecus 11.9±0.3	Estuarine Water	44	R 23	C	12		χ^2 = 0.243 p = 0.622	χ^2 = 0.091 p = 0.763	χ^2 = 9.091 p = 0.003
			L 21	T	32				
P. aztecus 11.5±0.2	Estuarine Water	50	R 30	C	17		χ^2 = 0.238 p = 0.626	χ^2 = 2.000 p = 0.157	χ^2 = 5.120 p = 0.024
			L 20	T	33				

[1] Numbers beneath species indicate 95% confidence intervals on mean total length.
[2] R and L indicate right and left sides of chamber
[3] C and T indicate treatment and control.

The two species responded differently to bromthymol blue (Table 1). Greater numbers of *P. setiferus* selected the stream containing the dye than the carrier water ($p = 0.0001$). Selection for the dye was significant and independent of selection for chamber side. *Penaeus aztecus* selected the dye and carrier water with approximately equal frequency in two experiments and those differences were not significant. Animal passage had no visible effect on dye flow stability.

Most postlarval movement was accomplished by rapid walking although some animals exhibited rapid upstream swimming. The maximum swimming velocity observed by both species was 70 $mm \cdot s^{-1}$. Most animals moved into the current after leaving the holding device. The proportion of shrimp which turned and moved with the current at least once prior to making a choice was 20% or less in all experiments (Table 2). Downstream movement did not preclude determination of a choice because observations continued until the animal had moved upstream and crossed the choice line beyond the central partition. This normally occurred within two minutes of release.

Regardless of the chamber side selected, the mean times to traverse the 21 cm distance to the choice line were similar for *P. aztecus* in the absence of dye (Table 2). The times required by *P. setiferus* to select both the chamber side and the dye or carrier were similar (Table 2). Differences were noted in the times required for *P. aztecus* to select the left or right side of the chamber when dye was present; however, no trend was apparent (Table 2) and differences were not significant ($F = 1.707, p = 0.170$). *Penaeus aztecus* took less time to select the dye than the carrier in both experiments (Table 2). In all experiments with both species the selection time was extremely variable among trials and the high variances associated with selection time rendered any differences among means nonsignificant.

Estuarine Water

Penaeus aztecus selected the estuarine water with a significantly greater frequency in all three experiments (Table 1). Selection for estuarine water was independent of selection for chamber side and no significant chamber bias was evident in these tests (Table 1). Mean selection times for the left and right sides of the chamber were lower in two trials (Table 2); however, the high variability in the data rendered differences non-significant ($F = 1.112, p = 0.357$). Similarly, the mean selection times for bay water were lower than those for carrier water in two of the experiments although no significant differences in selection times existed among the three experiments ($F = 0.663, p = 0.652$). The mean frequency of downstream movement during the estuarine water studies was significantly higher in the carrier water (control) stream than in the estuarine water ($t = 2.772, p = 0.025$).

TABLE 2
Mean selection times for chamber sides (left and right), treatments versus controls and the frequency of downstream movement in *P. aztecus* exposed to no stimulus, bromthymol blue and estuarine water, and in *P. setiferus* exposed to bromthymol blue.

Species	Treatment	n	Selection Time (s) Mean (Standard Deviation) Left		Right		Mean (Standard Deviation) Treatment		Control		Downstream Movement (%)[1]
P. aztecus	None	90	27.0	(31.2)	25.7	(35.5)	-		-		13.3
P. aztecus	None	60	37.5	(46.0)	37.9	(39.8)	-		-		15.0
P. setiferus	Bromthymol Blue	100	24.3	(26.2)	23.8	(24.2)	23.8	(25.6)	24.7	(24.1)	15.0
P. aztecus	Bromthymol Blue	57	18.8	(25.0)	29.0	(28.5)	19.3	(22.4)	29.4	(32.0)	14.0
P. aztecus	Bromthymol Blue	55	42.8	(38.3)	24.3	(23.5)	29.0	(30.7)	36.4	(34.1)	20.0
P. aztecus	Estuarine Water	50	103.4	(196.0)	36.5	(88.7)	57.4	(128.1)	115.4	(219.6)	18.0
P. aztecus	Estuarine Water	44	47.3	(94.0)	61.4	(104.3)	51.0	(97.9)	65.7	(104.7)	25.0
P. aztecus	Estuarine Water	50	51.7	(92.9)	125.7	(277.1)	115.9	(254.5)	57.6	(147.1)	24.0

[1]The proportion of trials in which an animal turned away from the upstream axis of the chamber by >90° at least once.

DISCUSSION

The results of the present experiments with and without a dye stimulus indicate that the chamber can provide unbiased estimates of the attraction or aversion to water quality. The gradual increase in the degree of mixing between the two streams appears to be a consequence of slight turbulence at the downstream edge of the partition coupled with head pressure induced by the resistance of the holding device. Studies with the glass chamber indicate that a narrower partition (2.7 mm) fitted with a sharper bevel (32°) provides superior separation of the two streams. Although the position of the animal holding device was fixed in the plexiglass chamber, the glass model incorporates a sliding holding device which can be positioned anywhere between the central partition and the downstream weir. This feature permits a longer exposure distance for faster moving species. The range of Nytex mesh sizes described in the plexiglass chamber may not be essential for good laminar flow as 400 μ mesh alone provided good laminarity in the glass chamber.

Significant chamber bias was encountered in early testing of the chamber and was attributed to uneven overhead illumination, opaque stains on the chamber sides and subtle deviations in the upstream alignment of the holding device.

Our design resolves several problems associated with conventional T- or Y-maze designs. Turbulent mixing of the two olfactants within the common region of the maze provides little information about the nature and geometry of the available choices. Animals face a sudden dichotomy where the maze arms diverge which forces a decision in a very short space while turbulence in the area of chamber divergence may provide conflicting information about olfactant geometry. Once an olfactant stream has been selected there is no opportunity to exit without reversing course and moving downstream. This may be a particular problem when working with positively rheotaxic species. Conventional Y-mazes do not maintain a constant olfactant concentration throughout the chamber. Concentrations usually drop below the junction of the maze arms. When working with two concentrations of the same olfactant, turbulent mixing in the common section of the chamber may result in a single intermediate concentration. The sharp delineation of both olfactant streams in our system permits precise positioning of the animal holding device along the interface of the streams. This provides each animal with information about available odor choices and their orientation prior to release. Animals are free to move through either olfactant between their release point and the central partition. The likelihood of accidentally selecting a less favorable olfactant is reduced by the increased opportunity to sample both choices. The ability to maintain constant and predictable exposure concentrations along the length of our chamber permits testing of two different olfactants or concentrations of the same olfactant, and overcomes the problems imposed by mixing in conventional chambers.

The use of Nytex mesh to promote laminar flow and air injection mixing were all adapted from choice chambers described in Souchek and Scarfe (1981) and Scarfe, *et al.* (1985). The latter study describes a system that provided up to eight separate olfactant streams which originated at the center of the chamber and flowed perpendicularly outward across an octagonal raceway. The incorporation of additional Nytex screens and a precisely machined bevel on the central partition have overcome the lateral dispersion which occurred in the octagonal chamber. Our chamber also differs with respect to the orientation of the olfactant streams relative to the raceway axis and the number of choices available to the animal. By orienting the olfactant stream boundary parallel to the current we hope to eliminate any potential conflicts between olfactant preference and rheotaxis. In addition, the difficulty of positioning an animal along the boundary of two olfactants while still orienting it along the raceway has been overcome in the present design. Although our chamber restricts testing to two olfactants and has fewer statistical degrees of freedom than the octagonal design, we have avoided the problems created by eight potential release points and the statistical difficulties associated with the dependent sequential observations of animal positions in the octagonal design.

The two species exhibited different responses to bromthymol blue dye. An olfactory basis for the preference exhibited by white shrimp cannot be ruled out; however, fundamental differences in the light/turbidity preferences and burrowing behaviors of the two species also suggest an explanation. *Penaeus setiferus* is generally regarded as a non-burrowing species which favors turbid water and is less sensitive to illumination than *P. aztecus* (Wickham and Minkler 1975). *Penaeus aztecus* burrows in response to bright light and is most active in very turbid water or at night (Wickham and Minkler 1975; Minello, Zimmerman and Martinez 1987). Burrowing by postlarvae in response to bright light has not been evaluated; however, investigations of the low temperature responses of brown and white postlarvae have suggested that *P. setiferus* does not burrow at this stage (Aldrich, Wood and Baxter 1968). Given their apparent inability to burrow, the dye selection recorded for *P. setiferus* may reflect a preference for the slightly lower light intensities which prevailed in the dye stream. Providing that the light intensities in both the dye and carrier water streams exceeded the burrowing threshold for *P. aztecus*, the lack of preference demonstrated by the species may have been caused by an attempt to seek a substrate favorable for burrowing. Use of the chamber in conjunction with bentonite suspensions may provide a mechanism for investigating the responses of these species to turbidity.

Both species were capable of the same maximum swimming velocity despite their size differentials. Hughes (1969, 1972) observed that postlarval *P. duorarum* which were similar in size to the brown shrimp we tested, were able to swim against a 5-5.5 $cm \cdot s^{-1}$ current in a circular chamber. Our observations suggest the utility of the chamber for evaluating sustained and burst postlarval swimming performance at higher current velocities.

The lower incidence of downstream movement suggests positive rheotaxis, a response which has not been documented in the postlarvae of either species. Positive rheotaxis to horizontal currents has been recorded in other decapod larvae such as the red king crab *Paralithodes camtschatica* (Shirley and Shirley 1988) and in the juvenile stages of pink shrimp *P. duorarum* (Fuss and Ogren 1966). Postlarval *P. duorarum* were positively rheotaxic in flood tides and negatively rheotaxis during ebb tides (Hughes 1972). This cyclical behavior was under tidal and endogenous control. In the present study, no relationship was noted between the incidence of downstream movement and the tidal phase which prevailed during each trial. The present results are biased by the consistent release of the postlarvae facing upstream and further studies which vary the release direction need to be undertaken in order to address the question of rheotaxis. The higher incidence of turning in the control stream suggests that animals which lost the odor stimulus moved downstream in an attempt to regain contact. In many instances this movement brought the postlarva into contact with the odor stream and a successful choice resulted.

The tests with Galveston Bay water confirm previous studies which suggested that estuarine water contains components which are attractive to shrimp (Kristensen 1964; Mair 1980). These results indicate the utility of the chamber for testing responses to natural olfactants. The salt marsh from which the water drained contains extensive stands of smooth cordgrass *Spartina alterniflora* which form an important nursery habitat for postlarval and juvenile brown shrimp. Any attraction to such habitats would be adaptive to immigrating postlarvae. Studies are underway to evaluate the attractiveness of *S. alterniflora* and other selected biological components of the marsh to both brown and white shrimp in order to identify the source(s) of the attractant in estuarine water.

ACKNOWLEDGMENTS

We thank the Fritz Chemical Company for supplying synthetic sea salts. A discussion with Dr. A.D. Scarfe influenced the initial form of the chamber and Drs. T.J. Minello and J. Lester provided statistical advice. Comments by Dr. J.W. Ambler and two anonymous reviewers improved this manuscript. This study was funded by a Texas A&M University at Galveston Research Enhancement Grant.

LITERATURE CITED

ALDRICH, D.V., C.E. WOOD and K.N. BAXTER. 1968. An ecological interpretation of low temperature responses in *Penaeus aztecus* and *P. setiferus* postlarvae. *Bulletin of Marine Science*. 18:438-443.

FORBES, A.T. and M.C. BENFIELD. 1986. Tidal behavior of post-larval penaeid prawns (Crustacea: Decapoda: Penaeidae) in a southeast African estuary. *Journal of Experimental Marine Biology and Ecology*. 102:23-34.

FUSS, C.M. and L.H. OGREN. 1966. Factors affecting activity and burrowing habits of the pink shrimp, *Penaeus duorarum* Burkenroad. *Biological Bulletin.* **130**:170-191.

HANSEN, D.J., S.C. SCHIMMEL and J.M. KELTNER, Jr. 1973. Avoidance of pesticides by grass shrimp (*Palaemonetes pugio*). *Bulletin of Environmental Contamination and Toxicology.* **9**:129-133.

HASLER, A.D., A.T. SCHOLZ and R.M. HORRAL. 1978. Olfactory imprinting and homing in salmon. *American Scientist.* **66**:347-355.

HUGHES, D.A. 1969. Responses to salinity change as a tidal transport mechanism of pink shrimp, *Penaeus duorarum. Biological Bulletin.* **136**:43-53.

_____. 1972. On the endogenous control of tide-associated displacements of pink shrimp, *Penaeus duorarum* Burkenroad. *Biological Bulletin.* **142**:271-280.

JOHNSEN, P.B. and A.D. HASLER. 1980. The use of chemical cues in the upstream migration of coho salmon, *Oncorhynchus kisutch* Walbaum. *Journal of Fish Biology.* **17**:67-73.

KEISER. R.K., Jr. and D.V. ALDRICH. 1973. A gradient apparatus for the study of salinity preference of small benthic and free swimming organisms. *Contributions in Marine Science.* **17**:153-162.

_____ and _____. 1976. Salinity preference of postlarval brown and white shrimp (*Penaeus aztecus* and *P. setiferus*) in gradient tanks. Texas A&M University Sea Grant College Program Report. TAMU-SG-75-208. 260 pp.

KRISTENSEN, I. 1964. Hypersaline bays as an environment of young fish. *Proceedings of the Gulf and Caribbean Fisheries Institute.* **16**:139-142.

MAIR, J.McD. 1980. Salinity and water-type preferences of four species of postlarval shrimp (*Penaeus*) from west Mexico. *Journal of Experimental Marine Biology and Ecology.* **45**:69-82.

_____, J.L. WATKINS and D.I. WILLIAMSON. 1982. Factors affecting the immigration of postlarval shrimp into a Mexican lagoon system. Proceedings of an International Symposium on Coastal Lagoons, SCOR/IABO/UNESCO, Bordeaux, France, 8-14 September, 1981. *Oceanology Acta.* **14(Supplement)**:339-345.

MAYNARD, D.J. and D.D. WEBER. 1981. Avoidance reactions of juvenile coho salmon (*Oncorhynchus kisutch*) to monocyclic aromatics. *Canadian Journal of Fisheries and Aquatic Sciences.* **38**:772-778.

MINELLO, T.J., R.J. ZIMMERMAN and E.X. MARTINEZ. 1987. Fish predation on juvenile brown shrimp, *Penaeus aztecus*: effects of turbidity and substratum on predation rates. *Fishery Bulletin.* **85**:59-70.

MOELENBERG, F. and T. KIOEBOE. 1973. Burrowing and avoidance behavior in marine organisms exposed to pesticide-contaminated sediment. *Marine Pollution Bulletin.* **14**:57-60.

ODUM, W.E. 1970. Insidious alteration of the estuarine environment. *Transactions of the American Fisheries Society.* **99**:836-847.

PENN, J.W. 1975. The influence of tidal cycles on the distributional pathway of *Penaeus latisulcatus* Kishinouye in Shark Bay, Western Australia. *Australian Journal of Marine and Freshwater Research.* **26**:93-102.

RUELLO, N.V. 1973. The influence of rainfall on the distribution and abundance of the school prawn (*Metapenaeus macleayi*) in the Hunter River region. *Marine Biology.* **23**:221-228.

SCARFE, A.D., C.W. STEEL and G.K. RIEKE. 1985. Quantitative chemobehavior of fish: an improved methodology. *Environmental Biology of Fishes.* **13**:183-194.

SOUCHEK, R.D. and A.D. SCARFE. 1981. A laminar flow chamber for behavioral studies of planktonic invertebrates and a sex pheromone study of the rotifer *Asplanchna brightwelli. The Biologist.* **63**:82-91.

SHIRLEY, S.M. and T.C. SHIRLEY. 1988. Behavior of red king crab larvae: phototaxis, geotaxis and rheotaxis. *Marine Behavior and Physiology.* **13**:369-388.

SORENSEN, P.W. 1986. Origins of the freshwater attractant(s) of migrating elvers of the American eel, *Anguilla rostrata. Environmental Biology of Fishes.* **17**:185-200.

STABELL, O.B. 1984. Homing and olfaction in salmonids: A critical review with special reference to the Atlantic salmon. *Biological Reviews.* **59**:333-388.

WICKHAM, D.A. and F.C. MINKLER. 1975. Laboratory observations on daily patterns of borrowing and locomotor activity of pink shrimp, *Penaeus duorarum*, brown shrimp, *Penaeus aztecus*, and white shrimp, *Penaeus setiferus*. *Contributions in Marine Science*. **19**:21-36.

Contributions in Marine Science. (1991) Vol. 32.

LONG-TERM TRENDS IN THE ABUNDANCE OF THE COPEPOD *ACARTIA TONSA* IN THE CALCASIEU ESTUARY

Michael Vecchione[1]

Department of Biological and Environmental Sciences
McNeese State University
Lake Charles, Louisiana 70609

ABSTRACT

Data on the abundance of adult *Acartia tonsa* are combined from three data sets. These data were all collected by similar methods at two stations in Calcasieu Lake and included samples taken during 51 months over a 132 month period (1975-1986). Both mean abundance and maximum abundance increased by a factor of 2 and then decreased to the original levels during this period. Although there was a slight trend of increasing abundance overall, this trend was not statistically significant. Analysis of a subset of these data would have led to very different conclusions. Water temperature decreased significantly and salinity increased significantly during this period, perhaps indicating increased circulation as dredged-spoil islands eroded. However, the relationship of *A. tonsa* abundance with temperature and salinity was not significantly different from random. All samples that contained high abundances (> 10000 individuals/m^3) were collected when the water column was not stratified in either salinity or temperature. This may indicate an interaction between wind-mixing of this very shallow water column and the vertical distribution of *A. tonsa*.

INTRODUCTION

Calcasieu estuary is located in southwest Louisiana. Its largest body of open water, Calcasieu Lake, covers approximately 255 km^2 to an average depth of 1-2 m and is separated from the Gulf of Mexico by the constriction of a narrow "pass" limiting circulation at its south end. This estuary has been subjected to major anthropogenic influences for more than a century (Gosselink, Cordes and Parsons 1979). Among these influences have been important hydrological alterations, human population growth, and the development of petroleum-production and petrochemical industries throughout the drainage basin. A persistent opinion among local residents is that environmental conditions have deteriorated substantially throughout the Calcasieu estuary during the past few decades and that this deterioration can be seen in decreased abundance of many animal species.

---

[1] Present address: NMFS Systematics Laboratory, National Museum of Natural History, Washington, DC 20560.

Gosselink, *et al.* (1979) summarized unpublished sources indicating annual production of *ca.* 6×10^6 bbls of crude oil and 43×10^{12} cf of natural gas in the Calcasieu basin. They listed 4×10^9 bbls of oil-well brine discharged annually to surface waters with annual phosphorus input of *ca.* 8×10^5 kg and assessed the eutrophic state of the basin as "dangerous." Other important pollutants include industrial organic toxins and the heavy metals; mercury, zinc, and copper; the oyster beds in Calcasieu Lake are closed each summer due to high numbers of coliform bacteria (Gosselink, *et al.* 1979). Densities of macrobenthic animals in Calcasieu Lake are lower than in other estuaries of the northern Gulf of Mexico (Gaston, Lee and Nasci 1988). Strong pollution impacts have been demonstrated in the bayous of the upper Calcasieu estuary for benthos (Gaston and Nasci 1988), nekton (Felley and Felley 1986; Felley 1987), and zooplankton (Vecchione 1989). Thus, the potential exists that the zooplankton in the lower estuary (Calcasieu Lake) suffer from man-made environmental stresses and that their populations have decreased as a result.

Temporal variability on a scale of years is an important natural component of zooplankton population dynamics (Haury, McGowan and Weibe 1978) that may be confused with anthropogenic changes. This component of variability is seldom assessed because of logistical difficulties (Likens 1983; Strayer 1986) but may be greater than the short-term temporal or small-scale spatial variability that is easier to study (Garrod and Colebrook 1978; Gieskes and Kraay 1977).

The calanoid copepod *A. tonsa* is the numerically dominant species of mesozooplankton in estuaries throughout southeastern North America and is seasonally dominant farther north. This species accounts for *ca.* 90% of the total number of copepods collected in the Calcasieu estuary, with abundances as high as 10^6 individuals per m^3 of water (Stubblefield, Lascara and Vecchione 1984). Closely related species of *Acartia* dominate the estuarine zooplankton in many parts of the world. Therefore, long-term changes in the abundance of *Acartia*, whether natural or anthropogenic, could have serious ecological repercussions.

This paper examines long-term patterns in the abundance of *A. tonsa* in Calcasieu Lake. Specifically, I wanted to determine the magnitude of interannual variability and whether any clear trend in the data that may be related to man's activities was apparent (*e.g.* consistent decrease in abundance over the period of a little more than a decade).

METHODS

The abundances presented here were derived from three separate studies of zooplankton in Calcasieu Lake. All three studies used very similar methods to collect zooplankton at two stations (identical in location in all three studies) in Calcasieu Lake over the 11 yr period 1975-1986. Data from two of these studies (Gillespie 1978; Vecchione, Lascara, Stubblefield and James 1983) have been reported separately, but not in the context of long-term variability.

All data reported here were collected during daytime at two widely-separated stations in Calcasieu Lake, one at the north end in Turner's Bay and the other at the south end near the "Old Jetties" (designated N1 and S1 in Fig. 1). These stations are listed in Gillespie (1978) as stations 1

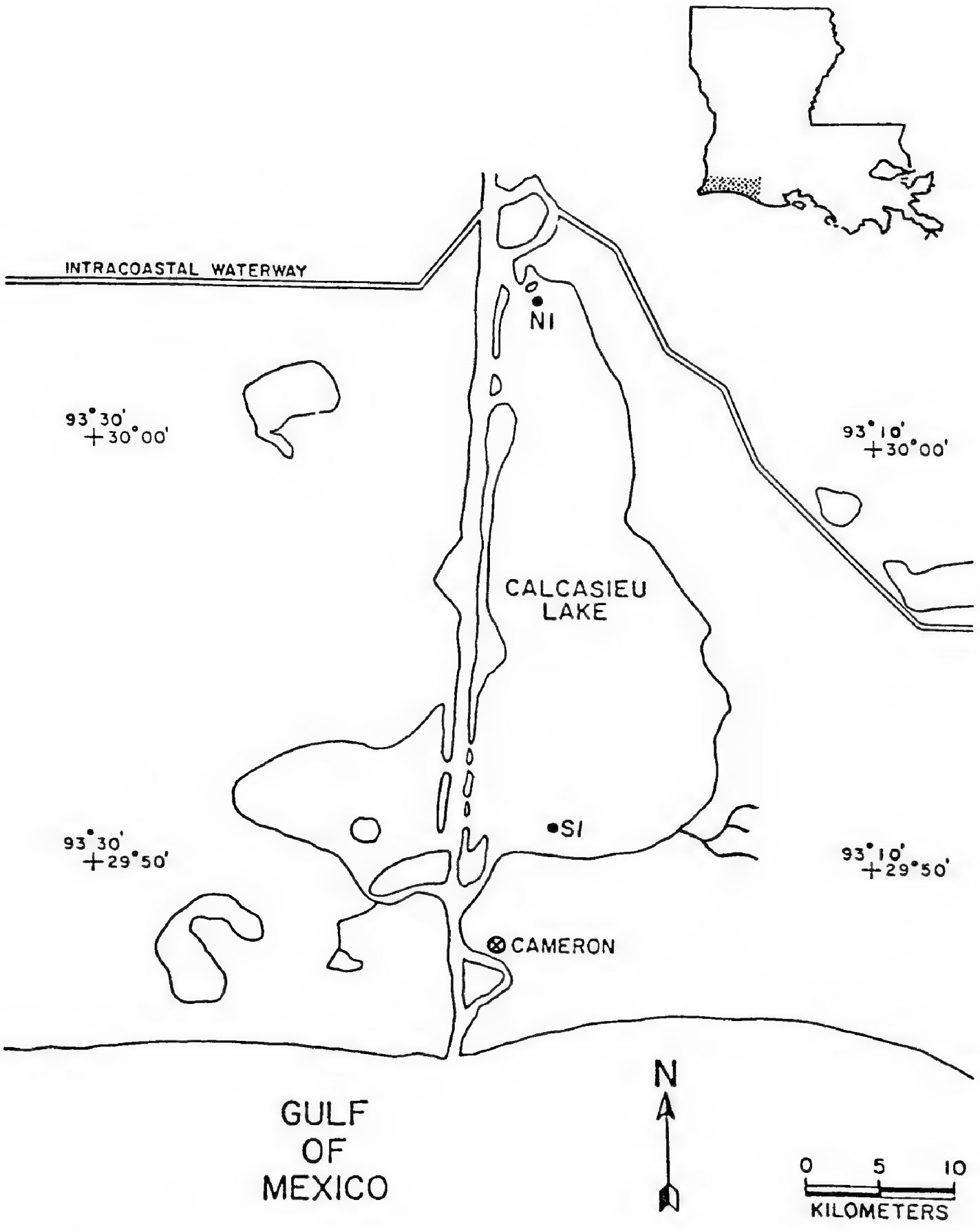


FIG. 1. Zooplankton sampling stations, both of which were used in three separate studies of Calcasieu Lake, southwest Louisiana.

and 2 of area VII, and in Vecchione, *et al.* (1983) as stations E2 and E4, respectively. All samples included in this presentation were collected using 0.5 m ring nets of 0.153 mm mesh netting. The first six months (October 1974 - March 1975) of Gillespie's (1978) study used a larger mesh size and therefore have not been included here. Thus, Gillespie's data presented here were collected monthly from April 1975 to September 1976. Vecchione, *et al.* (1983) also sampled monthly throughout the period February 1981 to October 1982. These stations have since been sampled on 18 occasions, although not on a regular monthly basis, from September 1983 to April 1986 (Felley, Vecchione and Hare 1987). These three data sets will be referred to below by the years in which sampling occurred (*i.e.*, 1975-76, 1981-82, and 1983-86).

Water temperatures and salinities were measured near the surface and near the bottom each time a station was sampled for all three data sets. These parameters were used to calculate indices of stratification of the water column. Stratification is defined here as surface salinity (or temperature) minus bottom salinity (or temperature).

Water at these stations is 1-2 m deep, so the nets had to be towed just below the surface to avoid hitting the bottom; duration of tow was one minute. The volume of water filtered on all tows was monitored using General Oceanics Model 2030 flowmeters. When sorting for copepods, subsampling was often necessary because of excessive abundance. Samples for the 1981-82 and 1983-86 data sets were split with a Folsom Plankton Splitter; those for the 1975-76 data set were subsampled "by volumetric aliquot" (Gillespie 1978). Specimens were characterized by major life-history stages as follows: nauplii, copepodids, adult males, or adult females. This paper reports on the abundance of total (males + females) adult *A. tonsa* as number of individuals collected per m^3 of water filtered for each sample.

Triplicate samples were collected during some of the 1981-82 and all of the 1983-86 trips. For purposes of comparison with the non-replicated stations, only the median value was used from triplicate sample sets. Variance among tows is presented elsewhere (Stubblefield, *et al.* 1984; Vecchione in preparation). Typically the Coefficient of Variation (CV = s.d./mean × 100) for the abundance of adult *A. tonsa* in triplicate samples is 100 or less, indicating that the distribution is not excessively patchy on the scale sampled by these one-minute tows.

Statistics presented here were computed using the Multivariate General Linear Hypothesis program in the SYSTAT statistical package for microcomputers.

RESULTS

In the combined data sets, samples were collected for 51 months of the 132-month period. The average abundance of adult *A. tonsa*, overall, was 4454 copepods/m^3. Although 75% of the samples collected < 5000 individuals/m^3, substantial variability existed in the samples that collected large numbers of *A. tonsa* (Fig. 2). In the samples with high abundance (> 10,000 individuals/m^3), abundances were much higher during the 1981-82 period than during either of the other periods. Samples with high abundance did not differ greatly between the 1975-76 period and the 1983-86 period. A similar pattern is apparent for mean abundance which during 1981-82 was 6844 individuals/m^3, compared with 2632 individuals/m^3 during 1975-76 and 3342 individuals/m^3 during 1983-86. Thus, the overall pattern in these data, in both maximum and mean abundance, is an approximate doubling followed by a decrease back to approximately the original level. It seems noteworthy that within pairs of successive years (*i.e.*, 1975-76; 1981-82; 1984-85) the variability in abundance is much less (Fig. 2). Based on linear-regression analysis, the trend in abundance over this 11-yr period shows a slight increase (slope =

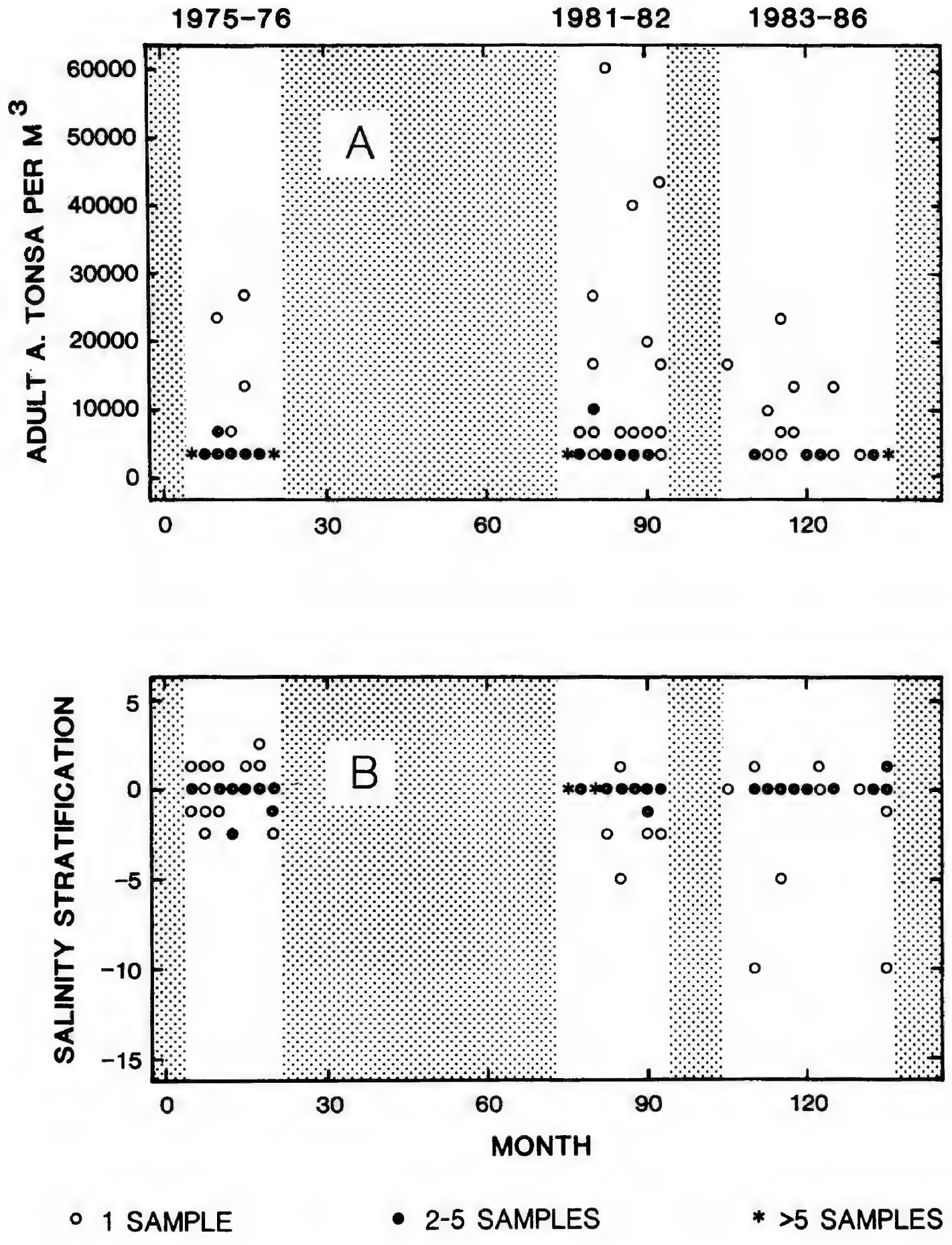


FIG. 2. Interannual variability in abundance of the copepod *Acartia tonsa* and water-column stratification in Calcasieu estuary. (A) Abundance of adult *Acartia tonsa* (individuals/m^3) plotted against chronological sequence of months starting on January 1975. (B) Stratification index δs (surface salinity minus bottom salinity) plotted against the same chronological sequence of months. Stippled areas represent periods for which no data are available.

17.2; standardized slope = 0.08) but is not significantly different from a null hypothesis of no change (t = 0.832, 2-tail p = 0.407, N = 103).

Linear-regression analysis indicated statistically significant changes in water temperature and salinity over the 11 yr period (Table 1).

TABLE 1.
Linear-regression analysis of water temperature and salinity in Calcasieu Lake over the 11-year period.

Parameter	Slope	Standardized Slope	t	p(2-tail)	N
Surface salinity	0.05	0.334	3.557	0.001	103
Bottom salinity	0.05	0.348	3.726	0.000	103
Surface temperature	-0.03	-0.119	-2.038	0.044	103
Bottom temperature	-0.03	-0.227	-2.341	0.021	103

Thus, water temperature decreased and salinity increased during this period (Table 2). A statistically significant negative correlation existed between temperature and salinity, both at the surface (Pearson's product-moment correlation r = $-0.235, p < 0.05$) and at the bottom (r = $-0.241, p <$ 0.05).

TABLE 2.
Changes in water temperature and salinity in Calcasieu Lake, based on data collected concurrently with zooplankton samples from two widely separated shallow (depth < 2 m) stations.

Salinity (X 10^{-3})

	Surface				Bottom			
Years	Min.	Mean	s.d.	Max.	Min.	Mean	s.d.	Max.
1975-76	0.1	10.4	5.1	18.5	0.1	11.2	5.5	20.7
1981-82	2.8	16.4	6.4	25.1	3.0	17.1	6.5	30.6
1983-86	0.0	15.4	6.9	23.3	4.7	16.8	6.5	30.9

Temperature (°C)

	Surface				Bottom			
Years	Min.	Mean	s.d.	Max.	Min.	Mean	s.d.	Max.
1975-76	10.8	23.0	6.1	31.3	10.8	23.3	6.1	31.5
1981-82	8.8	22.5	6.7	32.7	8.6	22.3	6.6	32.1
1983-86	4.7	18.5	7.7	29.1	4.8	18.5	7.7	29.4

Abundance was positively related to both temperature and salinity at both the surface and the bottom but these relationships were not statistically significant (Table 3). However, when abundance is compared with stratification of the water column, a clear pattern emerges (Fig. 3). All of the high abundances (> 10,000 individuals/m^3) occurred when δs, the difference in salinity between the surface and the bottom, was between −3 and 0; abundances > 20,000 individuals/m^3 were found only when δs was between −1 and 0. High abundances were similarly related to δt values (difference between surface and bottom temperature) between −1 and +1. Therefore, high abundances of adult *A. tonsa* were found only when the water column was not stratified in either temperature or salinity, although there were occasions when stratification was evident in one physical parameter but not in the other. Stratification in temperature was not significantly correlated with salinity stratification ($r = -0.022, p >> 0.05$). The pattern of changes in stratification in the long-term data set does not match that of the abundance of *A. tonsa* (Fig. 2).

DISCUSSION

Any sampling program is subject to limitations, but consistent sampling with fixed methods allows comparisons over a long time period. While considering these results, though, it is important to remember the limitations imposed by daytime net sampling for *A. tonsa*. Recent studies (Fulton 1984; Stubblefield, *et al.* 1984) have shown that *A. tonsa* aggregate near the bottom during daytime in shallow estuaries.

Thus, conventional net sampling for this species will seriously underestimate absolute abundance during the day. However, because these methods were used throughout the three studies, this would be a consistent bias and comparisons of relative abundance among years remains valid. I caution the reader, though, that these data estimate relative, rather than absolute, abundance.

Overall, the relative abundance of adult *A. tonsa* changed in this estuary by a factor of about 2 over a period of approximately a decade. The data also indicate that year-to-year variability in abundance was much less. The lack of a long-term downward trend in these data is not consistent with the hypothesis that ongoing environmental degradation is detrimental to the local population of this species. Indeed, the overall trend indicated a slight increase in abundance. The lack of statistical significance of this trend results from the high total variance in the data, variance that was deliberately included by monthly sampling at two geographically separated locations. However, given what may be a cyclic pattern of abundance, accurate determination of a trend independent of this cycle (and also determination of the period of this cycle) would require sampling over several of these decade-long fluctuations, an extremely long data base.

TABLE 3

Multiple linear-regression analysis of the relationship between abundance of *Acartia tonsa* (both raw data and after transformation by the natural logarithm) and the hydrographic parameters, bottom water temperature and salinity, at two shallow stations in Calcasieu estuary. Bottom hydrography is presented here because it correlated more strongly with *A. tonsa* abundance than did surface hydrography.

Untransformed Abundance Data

Variable	Coefficient	Std.Error	Std.Coeff	t	p(2-tail)
Salinity	-317.486	533.888	-0.229	-0.595	0.553
Temperature	-155.454	365.444	-0.114	-0.425	0.671
Interaction	24.415	23.178	0.428	1.053	0.295
Constant	4980.462	8800.542	0.000	0.566	0.573

Analysis of Variance

Source	Sum-of-Squares	df	Mean-Square	F-Ratio	p
Regression	4.19×10^9	3	1.40×10^9	1.638	0.185
Residual	8.43×10^{10}	99	8.52×10^8		

N = 103, Multiple r = 0.217, Adjusted Multiple $r^2 = 0.018$

Abundance Data Transformed by Natural Logarithm

Variable	Coefficient	Std.Error	Std.Coeff.	t	p(2-tail)
Salinity	0.059	0.115	0.193	0.509	0.612
Temperature	0.077	0.079	0.260	0.981	0.329
Interaction	0.000	0.005	-0.015	-0.038	0.970
Constant	4.292	1.900	0.000	2.259	0.026

Analysis of Variance

Source	Sum-of-Squares	df	Mean-Square	F-Ratio	p
Regression	31.048	3	10.349	2.608	0.056
Residual	392.903	99	3.969		

N-103, Multiple r = 0.271, Adjusted Multiple $r^2 = 0.045$

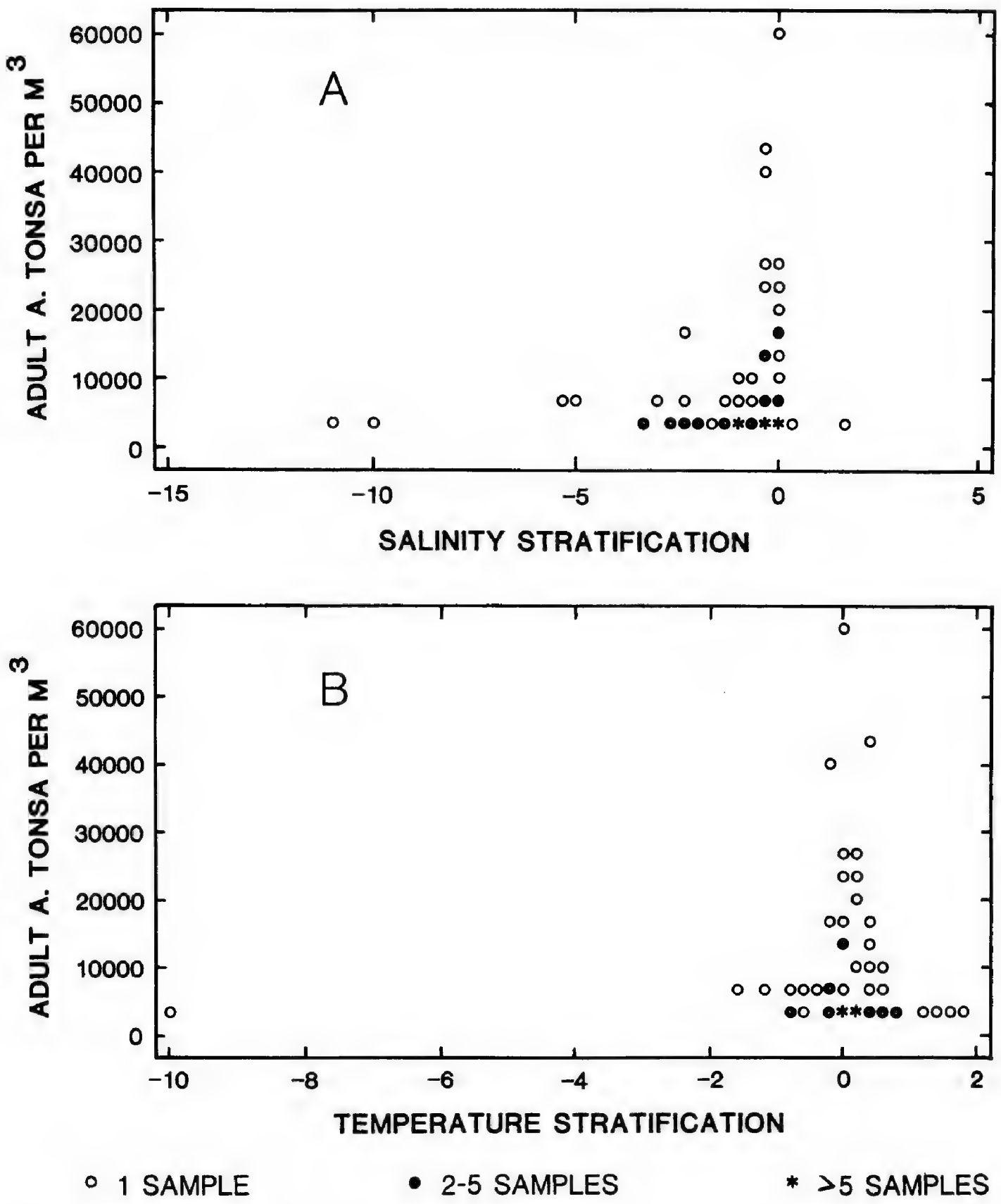


FIG. 3. Relationship of copepod abundance with water-column stratification in Calcasieu estuary. (A) Abundance of adult *Acartia tonsa* (individuals/m3) plotted against stratification index δs (surface salinity minus bottom salinity). (B) Abundance of adult *Acartia tonsa* (individuals/m3) plotted against stratification index δt (surface temperature minus bottom temperature).

Any interpretation based solely on subsets of these data would have resulted in very different inferences. For example, one of the goals of the 1983-86 study was interannual comparisons with the 1981-82 study. Exami-

nation of the data only from these two studies would lead one incorrectly to infer an alarming, precipitous, and statistically significant, decline in the abundance of *A. tonsa*. Similarly, the 1981-82 data were collected pursuant to a contract, one objective of which was comparison with the 1975-76 study. Comparison of these two studies would indicate a significant population explosion. Whereas either of these conclusions are supported by data and statistical inference, they fail to indicate the possibility of a long term cycle and the overall pattern of little or no net change seen over 11 years. In an environmental-impact study, such as the 1981-82 study, incomplete data and incorrect conclusions could lead to poor management decisions. Therefore, such studies should, wherever possible, include tasks that allow relative comparisons with previous studies for as long a temporal database as possible, even if this requires use of less-than-optimum methods.

Earlier zooplankton data do exist for these stations. Biases in these earlier data, unequal to the biases of the studies presented above, preclude direct comparisons with the data reported above, but it is appropriate to discuss the earlier studies here for the sake of comprehensiveness. Gillespie (1971) sampled the same stations during 1968-69, but used smaller nets with larger meshes. Thus, her calculated abundances for adult *A. tonsa*, which were much lower than those in any of the data sets presented above, may have resulted from avoidance of the small net or extrusion through the large mesh, rather than from population changes. Taken at face value, though, the 1968-69 data support a trend of increasing abundance over almost 20 years, as do the data from the first six months (October 1974-March 1975) of Gillespie's (1978) study, which were also omitted from the analysis above because of larger mesh size.

Dredging of the Calcasieu Ship Channel during the early 1960's resulted in dredged-spoil banks deposited on both sides of the channel, isolating Calcasieu Lake from sources of salt water and causing large changes in the local salinity regime (Gosselink, *et al.* 1979). These spoil banks have subsequently been breached, and natural erosion is being allowed to widen the openings. Improving circulation between Calcasieu Lake, where these stations were located, and the Calcasieu Ship Channel, which connects with the Gulf of Mexico, may have been responsible for the increase in salinity and decrease in temperature noted in this study. Of course, other phenomena, such as decreased rain fall in the watershed or industrial diversion of freshwater (Vecchione and Lascara 1983) may also explain the salinity changes. The report by Wiseman, Swenson and Power (1990) that salinity has decreased significantly at Calcasieu Pass in recent years is consistent with improving circulation between the lake and the ship channel.

It is possible that the relationship between peak abundance of *A. tonsa* and unstratified conditions is an artifact of sampling. The water column is not usually stratified in this shallow estuary, so a random distribution of very high abundances among these samples could produce a pattern similar to that found here. A formal test of this hypothesis would require selective sampling

under different stratification regimes while attempting to minimize temporal variability in abundance. However, three points argue against interpreting these data as a random relationship between abundance and stratification. First, thermal stratification was not correlated with salinity stratification. Second, several occasions were noted in which the water column was strongly stratified in one parameter but not in the other, indicating that vertical mixing was not strong enough to overcome stratification even though the water column was not stratified in one parameter; high abundances of *A. tonsa* were not found at these times. Third, high abundances of *A. tonsa* were found only when the water column was unstratified in both temperature and salinity. If the pattern described was simply an artifact of randomness, then high abundance could be expected on one or more occasions when the water column was stratified in one parameter but not the other.

The stations sampled for this study were shallow (< 2m) open-water areas that are strongly influenced by wind events (Stubblefield and Vecchione 1985; Vecchione, Maples and Donahoe 1986; Lee, Wiseman and Kelly 1990). Stratification is easily eliminated by turbulent mixing in these areas. *Acartia tonsa* tends to be most abundant very close to the bottom during the day in such shallow waters (Fulton 1984; Stubblefield, *et al.* 1984). The turbulence that eliminates stratification in these areas is also likely to stir the water column sufficiently that some of the near-bottom concentrations of *A. tonsa* are also mixed into the overlying water. Thus the relationship between high abundance of *A. tonsa* and lack of stratification may be a reflection of the interaction between wind-mixing and the vertical distribution of *A. tonsa*.

The strongest pattern in these data is the relationship between daytime abundance and water-column stratification. This may be considered to be a strong source of "noise" in the search for a long-term "signal" in the abundance data. However, because the interannual pattern of stratification does not match the pattern of *A. tonsa* abundance, I do not believe that the abundance pattern described above is simply an artifact of sampling in relation to wind events.

In summary, daytime sampling for adult *Acartia tonsa* by three similar studies over the 11 yr period 1975-1986 failed to demonstrate decreasing abundance, although analysis of a subset of these data would have indicated misleading temporal trends. During the period, abundance approximately doubled, then returned nearly to the original level. Water temperature decreased and salinity increased during this period. Very high abundances of *A. tonsa*, which aggregates near the bottom in daytime, were found only when the water column was not stratified in either temperature or salinity.

ACKNOWLEDGMENTS

I thank Cathy L. Stubblefield, Susan M. Felley, Sandra G.F. Hare, Cathy M. Lascara, and Kathy Conway for the many reasons that they already know. Martha Nizinski assisted with drafting.

George C. Grant, Thomas E. Bowman, and Austin B. Williams were kind enough to review a draft of the manuscript. This research was supported by Grant DE-FG01-83EP1111 from the U.S. Department of Energy.

LITERATURE CITED

FELLEY, J.D. 1987. Nekton assemblages of three tributaries to the Calcasieu estuary, Louisiana. *Estuaries*. **10**:321-329.

_____ and S.M. FELLEY. 1986. Habitat partitioning of fishes in an urban, estuarine bayou. *Estuaries*. **9**:208-218.

FELLEY, S.M., M. VECCHIONE and S.G.F. HARE. 1987. Incidence of ectoparasitic copepods on ichthyoplankton. *Copeia*. **1987**:778-782.

FULTON, R.S. 1984. Distribution and community structure of estuarine copepods. *Estuaries*. **7**:38-50.

GARROD, D.J. and J.M. COLEBROOK. 1978. Biological effects of variability in the North Atlantic Ocean. *Rapport et Procès-verbaux des Reunions. Conseil Permanent International Pour l'Exploration de la Mer*. **173**:128-144.

GASTON, G.R., D.L. LEE and J.C. NASCI. 1988. Estuarine macrobenthos in Calcasieu Lake, Louisiana: community and trophic structure. *Estuaries*. **11**:192-200.

_____ and J.C. NASCI. 1988. Trophic structure of macrobenthic communities in the Calcasieu estuary. *Estuaries*. **11**:201-211.

GIESKES, W.W.C. and G.W. KRAAY. 1977. Continuous plankton records: changes in the plankton of the North Sea and its eutrophic southern bight from 1948 to 1975. *Netherlands Journal of Sea Research*. **11**:334-367.

GILLESPIE, M.C. 1971. Analysis and treatment of zooplankton of estuarine waters of Louisiana. p. 109-175. In W.S. Perret, Cooperative Gulf of Mexico Estuarine Inventory and Study, Louisiana. Phase IV, Biology. Louisiana Wild Life and Fisheries Commission, New Orleans, Louisiana.

_____. 1978. Zooplankton analysis. *Louisiana Department of Wildlife and Fisheries Technical Bulletin*. **27**:27-80.

GOSSELINK, J.G., C.L. CORDES and J.W. PARSONS. 1979. An ecological characterization of the Chenier Plain coastal ecosystem of Louisiana and Texas. *U.S. Fish and Wildlife Service Office of Biological Services*. **FWS/OBS-78/9.**

HAURY, L.R., J.A. McGOWAN and P.H. WEIBE. 1978. Patterns and processes in the time-space scales of plankton distributions. pp. 277-327. In J.H. Steele (ed.), *Spatial Pattern in Plankton Communities*. Plenum Press, New York.

LEE, J.M., W.J. WISEMAN and F.J. KELLY. 1990. Barotropic, subtidal exchange between Calcasieu Lake and the Gulf of Mexico. *Estuaries*. **13**:258-264.

LIKENS, G.E. 1983. A priority for ecological research. *Bulletin of the Ecological Society of America*. **64**:234-243.

STRAYER, D. 1986. An essay on long-term ecological studies. *Bulletin of the Ecological Society of America*. **67**:271-274.

STUBBLEFIELD, C.L., C.M. LASCARA and M. VECCHIONE. 1984. The vertical distribution of zooplankton in a shallow turbid estuary. *Contributions in Marine Science*. **27**:93-104.

_____ and M. VECCHIONE. 1985. Zooplankton distribution in a wind-driven estuary before and after a major storm. *Contributions in Marine Science*. **28**:55-67.

VECCHIONE, M. 1989. Zooplankton distribution in three estuarine bayous with different types of anthropogenic influence. *Estuaries*. **12**:169-179.

_____ and C.M. LASCARA. 1983. A relationship between freshwater diversion and zooplankton community structure in a Chenier Plain estuary. *Estuaries*. **6**:289.

_____, _____, C.L. STUBBLEFIELD and W.O. JAMES. 1983. The relationship between brine-diffuser operation and zooplankton distribution. Ch. 9. In. L.R. DeRouen, *et al.*

(eds.), The West Hackberry Strategic Petroleum Reserve Site Brine Disposal Monitoring, Year I Report, Vol. III. McNeese State University, Lake Charles, Louisiana.

———————, R. MAPLES and R. DONAHOE. 1986. Changes in the distribution of chlorophyll *a* in a shallow water column during a solar eclipse. *Contributions in Marine Science.* 29:37-44.

WISEMAN, W.J., E.M. SWENSON and J. POWER. 1990. Salinity trends in Louisiana estuaries. *Estuaries.* 13:265-271.

Contributions in Marine Science. (1991) Vol. 32.

ICHTHYOPLANKTON OFF WEST LOUISIANA IN WINTER 1981-1982 AND ITS RELATIONSHIP WITH ZOOPLANKTON BIOMASS

James H. Cowan, Jr.[1] and Richard F. Shaw

Center for Wetland Resources
Louisiana State University
Baton Rouge, Louisiana 70803-7503

ABSTRACT

A total of 27,884 fish larvae was captured in 95 ichthyoplankton samples taken off west Louisiana during five mid-monthly cruises from December 1981 to April 1982. Fish larvae abundance and zooplankton biomass exhibited similar temporal and spatial trends (*i.e.*, an increase from January to April; higher values more shoreward in March and April) but correlations between larvae abundance and zooplankton biomass were seldom high for the 13 most commonly occurring taxa in months when their abundance peaked or was high. When the data were examined by station depth group, however, correlations were relatively high ($r \geq 0.30$) and significant on the mid- or outer-shelf (depth > 20 m), especially during the winter months (December to February). Interpretation of these data suggest that the seasonal presence of riverine water on the Louisiana shelf may be more important to larval fish during winter in more offshore waters when horizontal salinity gradients between inshore (riverine) and offshore (shelf) waters are steep and zooplankton biomass is generally lower, even well downstream of the source of the riverine input.

INTRODUCTION

Data from ichthyoplankton surveys provide valuable information on early life stages of fishes. Knowledge of seasonal occurrence, across-shelf distribution, and relative abundance of fish larvae in continental shelf waters is important for proper assessment and management of fish stocks, species identification, definition of spawning or nursery areas, and delineation of seasonal spawning migrations of adults (Saville 1964; Houde 1974; Colton, Smith, Kendall, Berrien and Fahay 1979; Ditty, Zieske and Shaw 1988).

Perhaps the most characteristic feature of the Louisiana continental shelf is the presence of two sources of freshwater discharge (*i.e.*, the Mississippi and Atchafalaya Rivers) that strongly influence the physics (Wiseman, Murray, Bane and Tubman 1982), biology (Hanson 1982; Wiseman, Turner, Kelly, Chuang, Rouse and Shaw 1986), and chemistry (Ho and Barrett 1975) of

[1] Present Address: The University of Maryland System, Center for Environmental & Estuarine Studies, Chesapeake Biological Laboratory, P.O. Box 38, Solomons, Maryland, 20688-0038.

northwestern Gulf of Mexico (Gulf) waters. The freshwater runoff, averaging 1.83×10^4 m^3s^{-1} with lowest discharges in winter and highest in spring (Dinnel and Wiseman 1986), contributes to a low salinity, highly turbid, nearshore water mass of coastal boundary layer (CBL) inside a steep oceanic salinity front on the inner- to mid-shelf about 20 to 50 km offshore (Crout 1983; Cochrane and Kelly 1986).

Although frontal boundaries in shelf seas have received considerable attention (Iles and Sinclair 1982; Mackas, Denman and Abbott 1985; Sakamoto and Tanaka 1986; Smetacek 1986; Townsend, Christensen, Stevenson, Graham and Chenoweth 1987; Wolanski and Hamner 1957), the effects of riverine plumes and fronts on the distribution of fish larvae remains unclear (Govoni, Hoss and Colby 1989). In recent years, studies of the Mississippi River plume, front and adjacent northern Gulf of Mexico waters have produced discrepant results about the influence of these waters on the Louisiana shelf's ichthyoplankton distribution and feeding ecology (Shaw, Cowan and Tillman 1985a; Shaw, Wiseman, Turner, Rouse, Chuang and Kelly 1985b; Govoni, Chester, Hoss and Ortner 1985; Sogard, Hoss and Govoni 1987; Cowan and Shaw 1988; Govoni, *et al.* 1989; Govoni and Chester, in press). In part, these discrepancies may result from the dynamic and ephemeral nature of the frontal boundaries in the northern Gulf (Govoni, *et al.* 1989), especially near the source of the freshwater input.

Wiseman, *et al.* (1986) described several biological and chemical associations along a frontal boundary in the northwestern Gulf west of the Atchafalaya River Delta in the winter and early spring of 1981-82. They showed that high concentrations of nutrients (primarily nitrates and silicates) were introduced with riverine water on the shelf, producing areas of high primary productivity and elevated zooplankton biomass progressively seaward of the offshore edge of the CBL, peaking on the mid-shelf. From December to April river discharge, frontal salinity gradients, nutrient input, and zooplankton biomass generally increased while salinity decreased; changes between February and March were dramatic (Wiseman, *et al.* 1986).

In this paper, we present survey data on the distribution and seasonality of the continental shelf ichthyoplankton off west Louisiana in the winter and early spring of 1981-82. This information supplements previous shelf studies and reviews (*e.g.*, Finucane, Collins and McEachran 1977; Ditty, *et al.* 1988). We also have used the relative abundance (larvae m^{-2}) of the most abundant (> 1% of total catch) ichthyoplankton taxa collected concurrently with zooplankton data, to examine the correlative relationship between fish larvae distribution (and abundance) and zooplankton biomass (concentration in mg m^{-3}) in west Louisiana shelf waters influenced by significant riverine input.

METHODS

Ichthyoplankton samples (N = 95) used in this analysis were collected from 24 stations on 3 transects from west of Sabine Pass, Texas (SP) to east of the Mermantau River, Louisiana (MR)

(Fig. 1). The outer or longer transects extended to the shelf break (200 km). The middle or short transect extended to the 18-m depth contour and was approximately 65 km long.

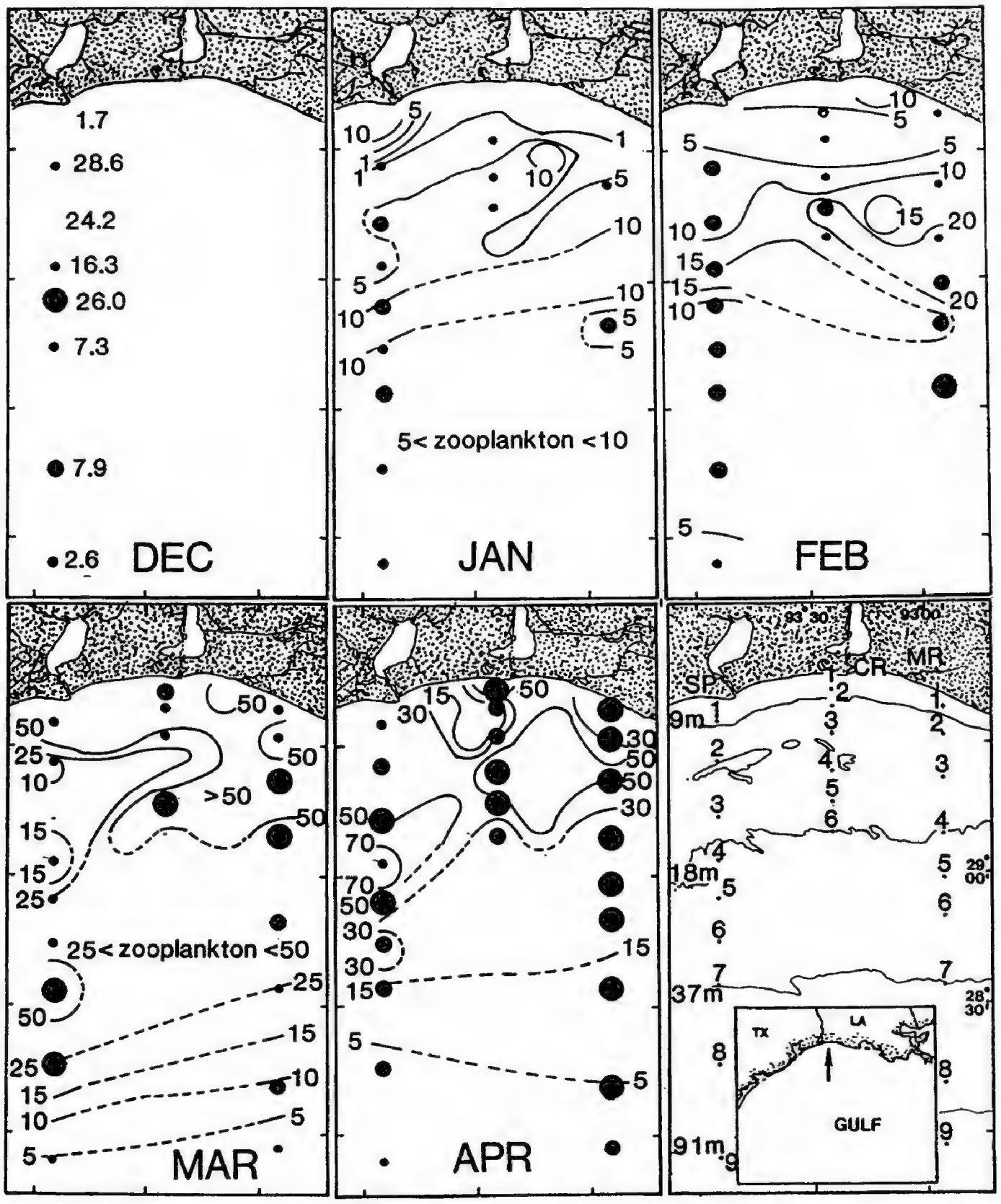


FIG. 1. Total abundance (larvae m^{-2}) of fish larvae collected off west Louisiana by month, December 1981 - April 1982. Abundances are as follows: 0 = no larvae collected; (·) = > 0-10; (·) = > 10-35; (•) = > 35-75; (●) = > 75-150; and (●) = > 150 larvae m^{-2}. Also shown are station grid and selected isobaths. SP = Sabine Pass, TX; CR = Calcasieu River, LA; MR = Mermentau River, LA. Contour lines are zooplankton concentration (mg m^{-3}).

Samples were collected during mid-monthly cruises from December 1981 to April 1982. An opening and closing, 60-cm, bongo-type plankton sampler (net mesh 335 μm) was to make 10-min stepped oblique tows from near bottom (or 200 m if station depth > 200 m) to surface at each station. A General Oceanics flowmeter (Model No. 2030) was mounted in the mouth of each net to quantify the volume of water filtered. Zooplankton biomass (mg m^{-3}) estimates were made using aliquot dry weights after nonplanktonic organisms and flotsam were removed from samples. Ichthyoplankton collections were reported as relative abundance at a station (larvae m^{-2} sea surface). Data on larvae presented here are primarily for individuals < 10 mm total length (TL). Information on some large larvae (*i.e.*, leptocephali of Nettastomatidae, Congridae and Ophichthidae), as well as all other larvae collected also are included (Table 1). Pearson Product-Moment correlation coefficients (r) were obtained from the relationship between larvae abundance and zooplankton biomass at a station or station groups (PROC CORR, SAS Institute, Inc. 1985).

RESULTS

A total of 27,884 fish larvae was collected in the samples taken from the three transects off west Louisiana (Table 1). Seventy-two species in 44 families were identified; nineteen genera for which species were not identified also were included. Members of six families comprised 85.4% of the total catch (23,821 larvae). These families, ranked by abundance and given with number of individuals and number of species, respectively, were Clupeidae, 9,430: 3 species; Engraulidae, 8,609: 4 species; Sciaenidae, 2,387: 6 species; Bregmacerotidae, 1,219: 3 species; Synodontidae, 1,101: 3 species; and Gobiidae, 1,075: 3 species.

Overall, abundance of larvae was highest in April (73.4 ± 73.8 larvae m^{-2}). Abundance in December, January, February and March was 38.1 ± 49.4, 14.3 ± 19.1, 35.7 ± 46.8 and 31.8 ± 50.4 larvae m^{-2}. Thirteen species occurred in abundances that exceeded 1% of the total catch (Table 2). Together, these species represent 63% (17,600 individuals) of all larvae collected. With the exception of the clupeids and anchovies, and to a lesser extent *Cynoscion arenarius*, monthly mean abundance was relatively constant, with a slight increase from January to April. The abundances in December must be viewed in consideration of an abbreviated cruise track due to adverse weather conditions which allowed only the western-most, long transect to be sampled. Consequently, the total number of stations and water volume filtered was less in December than in other months.

In general, larvae were more abundant over the mid-shelf (> 20 m depth) in December to February (Fig. 1). By March and April, highest abundances were found inshore of mid-shelf and in the eastern portion of the sampling grid, even though larvae were collected at all stations. Species whose larvae peaked in abundance from December to February occurred primarily at stations with depths > 20 m with few larvae being collected at stations with depths < 10 m. By April, most larvae were collected at stations with depth of 10-37 m (Fig. 1).

TABLE 1
Taxa, total number and months of occurrence of larval fish collected in west Louisiana shelf waters in winter and early spring 1981-82.

TAXA	TOTAL NUMBER	MONTHS OF OCCURRENCE
Elopidae		
Elops saurus	3	Jan; Apr
Nettastomatidae		
Hoplunnis sp.	8	Dec; Mar - Apr
Congridae	2	Jan
Ophichthidae		
Myrophis punctatus	17	Dec - Feb
Ophichthus sp.	6	Mar - Apr
Clupeidae	211	Jan - Apr
Brevoortia patronus	6,509	Dec - Apr
Etrumeus teres	2,646	Dec - Apr
Harengula jaguana	64	Mar - Apr
Engraulidae	6,006	Apr
Anchoa hepsetus	1,432	Apr
Anchoa mitchilli	800	Feb - Apr
Anchoa nasuta	1	Apr
Anchoviella perfasciata	370	Feb - Apr
Argentinidae		
Argentina silus	1	Feb
Gonostomatidae	1	Mar
Vinciguerria poweriae	8	Apr
Vinciguerria sp.	1	Apr
Cyclothone braueri	7	Feb - Mar
Maurolicus muelleri	242	Feb - Apr
Synodontidae	658	Dec - Apr
Saurida brasiliensis	2	Dec; Mar
Synodus foetens	438	Feb - Mar
Synodus saurus	3	Mar

Table 1 - continued

Paralepidae	3	Mar
Lestidiops affinis	2	Mar
Lestrolepis intermedia	2	Apr
Myctophidae	43	Dec - Apr
Hygophum hygomii	71	Apr
Hygophum reinhardtii	1	Apr
Myctophum nitidulum	1	Apr
Myctophum asperum	5	Apr
Myctophum obtusirostre	1	Feb
Myctophum selenops	3	Feb
Myctophum sp.	22	Feb - Apr
Diogenichthys atlanticus	1	Mar
Benthosema glaciale	1	Mar
Benthosema suborbitale	4	Jan - Feb
Ceratoscopelus warmingi	33	Feb - Apr
Lampanyctus sp.	6	Mar
Diaphus rafinesquii	117	Feb - Apr
Diaphus sp.	21	Mar - Apr
Centrobranchus chaeracephalus	1	Jan
Omosudidae		
Omosudis lowei	1	Feb
Antennariidae		
Antennarius radiosus	3	Apr
Histrio histrio	1	Feb
Gobiesocidae		
Gobiesox strumosus	9	Apr
Bregmacerotidae		
Bregmaceros atlanticus	10	Jan - Apr
Bregmaceros macclellandi	1	Jan
Bregmaceros cantori	1,208	Dec - Apr
Scopelarchidae		
Scopelarchus quentheri	1	Feb

Table 1 - continued

Gadidae		
Urophycis floridana	14	Jan - Mar
Urophycis sp.	9	Dec - Mar
Ophidiidae		
Brotula barbata	1	Mar
Ophidion sp. Type I	4	Dec - Jan
Ophidion sp. Type II	1	Jan
Atherinidae		
Membras martinica	5	Feb
Centriscidae		
Macrorhamphosus scolopax	7	Feb - Mar
Syngnathidae		
Syngnathus elucens	1	Apr
Syngnathus louisianae	12	Jan - Apr
Serranidae	5	Feb
Centropristis philadelphica	238	Jan - apr
Diplectrum sp.	3	Mar
Epinephelus sp.	16	Apr
Hemanthias leptus	115	Dec; Apr
Serranus sp.	131	Dec - Apr
Malacanthidae		
Caulolatilus sp.	22	Apr
Carangidae		
Trachurus lathami	117	Jan - Apr
Coryphaenidae		
Coryphaena hippurus	1	Apr
Haemulidae	23	Apr
Sparidae	54	Jan - Mar
Calamus sp.	271	Dec - Apr
Lagodon rhomboides	329	Jan - Apr

Table 1 - continued

Sciaenidae		
Cynoscion arenarius	1,508	Jan - Apr
Larimus fasciatus	8	Apr
Leiostomus xanthurus	252	Dec - Mar
Menticirrhus americanus	41	Apr
Menticirrhus sp.	79	Apr
Micropogonias undulatus	463	Dec - Apr
Pogonias cromis	36	Jan - Apr
Mugilidae		
Mugil sp.	52	Dec - Apr
Sphyraenidae		
Sphyraena barracuda	2	Apr
Blenniidae	33	Jan; Mar - Apr
Hypsoblennius sp.	8	Mar
Eleotridae	4	Apr
Gobiidae	3	Jan - Mar
Bollmannia communis	503	Dec - Apr
Gobionellus sp.	566	Dec - Apr
Gobiosoma sp.	3	Mar - Apr
Gempylidae	1	Mar
Trichiuridae		
Trichiurus lepturus	7	Apr
Scombridae	5	Dec; Mar - Apr
Auxis I (Fahay 1983)	2	Jan
Auxis sp.	1	Mar
Scomber japonicus	2	Mar
Scomberomorus cavalla	13	Mar - Apr
Scomberomorus maculatus	6	Apr
Thunnus albacares	1	Apr
Thunnus thynnus	4	Apr

Table 1 - continued

Stromateidae	17	Feb - Apr
Peprilus alepidotus	8	Apr
Peprilus burti	714	Dec - Apr
Scorpaenidae		
Scorpaena sp.	34	Mar - Apr
Triglidae	119	Dec - Apr
Prionotus sp.	159	Dec - Apr
Dactylopteridae		
Dactylopterus volitans	19	Dec - Apr
Bothidae	46	Dec - Apr
Bothus sp.	7	Dec
Citharichthys cornutus	13	Mar
Citharichthys spilopterus	84	Jan - Apr
Engyophrys senta	6	Apr
Etropus crossotus	42	Mar - Apr
Etropus sp.	484	Dec - Apr
Paralichthys lethostigma	27	Jan - Apr
Trichopsetta ventralis	9	Apr
Pleuronectidae	1	Feb
Cynoglossidae		
Symphurus sp.	80	Dec - Apr
Tetraodontidae		
Sphoeroides sp.	24	Apr
	27,884	

TABLE 2

Monthly abundance (± standard deviation) and total number of the most frequently occurring taxa of larval fish collected in west Louisiana continental shelf waters in winter and early spring 1981-82. Number in parenthesis is the correlation (Pearson Product-Moment) between larvae abundance (number m^{-2}) and zooplankton biomass (mg m^{-3}). N = number of stations used in the correlation analysis for each species.

TAXA	ABUNDANCE (larvae m^{-2} sea surface)					TOTAL NUMBER
	DECEMBER (N=8)	JANUARY (N=20)	FEBRUARY (N=21)	MARCH (N=22)	APRIL (N=24)	
Brevoortia patronus	0.7 ± 1.1 (0.64)	3.2 ± 5.4 (0.72)*	24.6 ± 44.3 (0.22)*	7.6 ± 21.9 (-0.11)	0.01 ± 0.03 (0.54)	6,509
Etrumeus teres	6.8 ± 14.3 (0.77)*	2.9 ± 8.6 (-0.07)	4.7 ± 10.8 (-0.04)	4.0 ± 10.0 (0.15)	2.1 ± 5.0 ((-0.38)	2,6461
Cynoscion arenarius	0 -	1 larva -	0.1 ± 0.2 (0.27)	0.4 ± 0.6 (0.46)*	3.8 ± 5.7 (0.46)*	1,508
Anchoa hepsetus	0 -	0 -	0 -	0 -	6.3 ± 6.5 (0.36)*	1,432
Bregmacerous cantori	10.3 ± 26.1 (-0.10)	1.1 ± 4.5 (0.06)	0.44 ± 13.1 (-0.20)	4.1 ± 13.1 (-0.19)	4.3 ± 9.4 (-0.37)	1,208
Anchoa mitchilli	0 -	0 -	0 -	1 larva -	2.8 ± 3.7 (0.25)	800
Peprilus burti	3.7 ± 6.1 (0.79)*	0.8 ± 1.9 (0.25)	1.0 ± 2.1 (0.02)	0.7 ± 1.4 (-0.22)	0.3 ± 0.5 (-0.02)	714

Table 2 - continued

Gobionellus sp.	4.6 ± 6.1 (0.18)	0.2 ± 0.5 (-0.04)	0.4 ± 0.8 (-0.14)	0.7 ± 1.2 (-0.38)	1.9 ± 7.9 (-0.15)	566
Bollmannia communis	0.7 ± 1.8 (0.05)	0.1 ± 0.4 (-0.03)	0.1 ± 0.2 (0.01)	1.6 ± 4.4 (-0.15)	2.7 ± 6.0 (-0.34)	503
Etropus sp.	1.1 ± 1.8 (-0.02)	0.4 ± 1.6 (0.16)	0.5 ± 2.0 (0.002)	0.4 ± 1.1 (-0.30)	1.9 ± 3.2 (0.20)	484
Micropogonias undulatus	0.5 ± 0.9 (0.07)	0.6 ± 1.4 (0.16)	0.2 ± 0.4 (-0.02)	0.3 ± 1.2 (0.17)	1 larva -	463
Synodus foetens	0 -	0 -	0.2 ± 0.6 (-0.08)	2.0 ± 5.7 (-0.04)	2.1 ± 6.3 (-0.18)	438
Lagodon rhomboides	0 -	0.1 ± 0.2 (0.14)	0.2 ± 0.6 (-0.02)	2.9 ± 8.6 (0.18)	0.04 ± 0.1 (0.25)	329
Zooplankton (mg m^{-3})	2.0 ± 0.7	1.8 ± 0.4	1.9 ± 0.3	2.3 ± 1.0	2.3 ± 0.9	
TOTAL LARVAE	38.1 ± 49.4 (0.36)*	14.3 ± 19.1 (0.35)*	35.7 ± 46.8 (0.19)*	31.8 ± 50.4 (-0.08)	73.4 ± 73.8 (-0.09)	

*statistically significant ($p < 0.05$)

Monthly mean zooplankton biomass also increased from January to April (Table 2). In December to February, highest zooplankton biomass (10-25 mg m^{-3}) occurred at stations on the inner to mid-shelf (Fig. 1). Zooplankton concentrations inshore (< 10 m) and in offshore waters (> 70 m) were relatively low (1-10 mg m^{-3}). Zooplankton biomass increased dramatically in March and April (15 to 70 mg m^{-3}) with higher values being found from inshore to the outer stations of the mid-shelf (Fig. 1). Zooplankton biomass in offshore waters > 70 m deep remained relatively low (usually < 10 mg m^{-3}).

Despite the similar temporal and spatial trends in total fish larvae abundance and zooplankton biomass (*i.e.*, increase from January to April; higher values more shoreward in March and April) correlation coefficients between mean larvae abundance and zooplankton biomass at a station were seldom high or significant for the 13 most abundant ichthyoplankton taxa in months when their abundance peaked or was high (Table 2). When the data were examined by station depth group, however, a stronger correspondence was evident. For most taxa that peaked in abundance from December to February, correlations between mean larvae abundance and zooplankton biomass (Table 3) often were relatively high and significant on the mid- or outer-shelf (20-75 m) with the exceptions of *Peprilus burti* and *Micropogonias undulatus*. Correlations also were high on the mid- to outer shelf for some ichthyoplankton taxa that peaked in abundance in March or April (Table 3). These results indicate a significant correlation between larvae abundance and zooplankton biomass, especially in the early winter on the mid- and outer-shelf seaward of the CBL.

DISCUSSION

Whether the correlations between larvae abundance and zooplankton biomass (Tables 2 and 3) were a result of biological interaction with the zooplankton (food availability for larvae resulting in increased growth and reduced mortality) or physical processes (Govoni, *et al.* 1989) is not known. Data available here were for macrozooplankton (*i.e.*, collected in 335 μm net meshes; primarily adult copepods). Microzooplankton concentration at the size pertinent to feeding larvae (50-200 μm) was not adequately sampled. However, Al-Yamani (1988) reported positive correlations between fish larvae, adult copepods and copepod nauplii in Mississippi River plume and adjacent waters during February 1980, 1981 and 1982 and Dagg, Ortner and Al-Yamani (1988) showed that maximum concentrations (~ 200 L^{-1}) of copepod early life stages occurred downstream of Mississippi River plume waters (*i.e.*, west of the river's confluence with the Gulf) in the winter of 1981-82. Several other studies have reported that waters in and adjacent to the plume often represented good feeding areas for larval fish (Dagg, *et al.* 1988; Al-Yamani 1988; Finucane, Grimes and Naughton 1990; Govoni and Chester

TABLE 3

Across-shelf (station depth) distribution (abundance ± standard deviation) of the thirteen most abundant taxa of larval fish collected in west Louisiana shelf waters in winter and early spring 1981-82. Number in parenthesis is the correlation (Pearson Product-Moment) between larvae abundance (number m^{-2}) and zooplankton biomass (mg m^{-3}). N = number of stations used in the correlation analysis for each species.

TAXA	DEPTH (m) (Shelf Location)				Peak Seasonal Occurrence
	<10 (N=18) (Inshore)	10-35 (N=49) (Mid-Shelf)	35-70 (N=19) (Mid-Shelf)	>70 (N=9) (Outer Shelf)	
Etrumeus teres	0.01 ± 0.04 (-0.27)	2.9 ± 8.9 (0.10)	21.1 ± 19.5 (0.51)*	5.0 ± 5.5 (0.36)	Dec
Bregmacerous cantori	0 -	0.3 ± 1.1 (0.01)	20.3 ± 27.7 (0.20)*	11.8 ± 12.8 (0.22)*	Dec
Peprilus burti	0.03 ± 0.1 (-0.18)	1.0 ± 2.7 (0.24)	2.8 ± 2.7 (-0.04)	0.5 ± 0.7 (0.02)	Dec
Gobionellus sp	0.02 ± 0.04 (0.10)	0.3 ± 1.2 (0.29)	2.8 ± 5.2 (-0.04)	7.2 ± 12.2 (0.47)*	Dec
Micropogonias undulatus	0.4 ± 1.4 (0.13)	0.3 ± 0.8 (-0.11)	0.5 ± 1.3 (0.07)	0.1 ± 0.3 (0.10)	Jan
Brevoortia patronus	1.2 ± 2.0 (0.43)*	12.3 ± 30.8 (0.38)*	0.5 ± 1.4 (0.07)	0 -	Feb
Lagodon rhomboides	0.01 ± 0.04 (0.23)	0.1 ± 0.3 (-0.07)	6.9 ± 12.8 (0.59)*	0.3 ± 0.8 (-0.29)	Mar

Table 3 - continued

Cynoscion arenarius	0.6 ± 1.2 (0.66)	1.2 ± 4.1 (0.49)*	0.1 ± 0.1 (0.59)*	0.04 ± 0.1 (0.39)*	Apr
Anchoa hepsetus	0.6 ± 1.2 (0.20)	2.4 ± 5.2 (0.45)*	0 -	0 -	Apr
Anchoa mitchilli	0.7 ± 2.1 (-0.14)	0.9 ± 2.5 (0.46)*	0 -	0 -	Apr
Bollmannia communis	0.004 ± 0.2 (-0.18)	0.3 ± 0.8 (0.08)	6.0 ± 7.6 (-0.19)	4.5 ± 7.8 (0.25)	Apr
Etropus sp.	0.01 ± 0.02 (0.36)	0.8 ± 2.2 (0.35)	2.9 ± 3.4 (-0.06)	0.9 ± 1.5 (0.02)	Apr
Synodus foetens	0 -	0.1 ± 0.4 (0.05)	6.8 ±10.1 (-0.12)	3.5 ± 7.6 (0.30)*	Apr
Zooplankton (mg m^{-3})	2.3 ± 1.2	2.1 ± 0.5	2.0 ± 0.8	1.4 ± 0.3	Mar-Apr
TOTAL LARVAE	7.2 ± 12.8 (0.16)	34.7 ± 39.8 (0.51)*	97.9 ± 53.9 (0.26)*	83.1 ± 113.1 (0.35)*	Apr

*statistically significant ($P < 0.05$)

1990) and larvae abundances and/or densities can be high there (Sogard, *et al.* 1987; Govoni, *et al.* 1989; Shaw and Drullinger 1990a, b).

It is likely that the abundance and distribution of fish larvae on the west Louisiana shelf (within the sampling grid; Fig. 1) is seasonally influenced by the presence of nutrient-rich freshwater issuing from the Mississippi and Atchafalaya Rivers. These rivers reach maximum discharge in April (on average). Wiseman, *et al.* (1986) showed several strong biological and chemical associations that apparently resulted from the riverine water, and that acted to influence zooplankton distribution and biomass along the Louisiana coast well downstream (*i.e.*, west ~ 175 km) of the Atchafalaya River. Conservative parameters (*e.g.*, salinity) have been used to trace the influence of the Atchafalaya and Mississippi Rivers on the inner shelf westward to Corpus Christi Bay, Texas (Angelovic 1976; Temple, Harrington and Martin 1977; Cochrane and Kelly 1986). The annual discharge from these rivers represents approximately 10% of the volume of the west Louisiana-Texas shelf. Fresh-water from the Mississippi River could reach the Mexican border in as little as two months assuming a 15 cm sec^{-1} average water current speed in the inner-shelf band (Dinnel and Wiseman 1986).

Interpretation of our plankton data suggests that the influence of plume or CBL waters on the distribution (and perhaps survival) of recently-spawned larval fish also may be more important during winter months when horizontal salinity gradients between inshore (riverine) and offshore (shelf) waters are steep, the CBL is well defined, and the area of maximum freshwater content has been displaced offshore (Fig. 2) (Cochrane and Kelly 1986; Dinnel and Wiseman 1986; Wiseman, *et al.* 1986 for review). This may be especially true during early winter (December to February) when zooplankton concentrations are generally lower (Fig. 1). Also, the winter time period, with its high frequency (3-8 day intervals) of atmospheric cold front passages and winds from the north, may provide a mechanism for the periodic offshore transport of nutrients and phytoplankton from surface waters of the CBL to the inner and mid-shelf regions resulting in increased zooplankton biomass (Dagg 1988). The modal lengths of *Brevoortia patronus* and *Etrumeus teres*, most abundant in December to February on the mid-shelf at stations with higher zooplankton biomass, was ~ 5 to 7 mm TL and suggests a relationship between small larvae of these winter spawners and zooplankton food (Table 3). Even correlations between total larvae abundance for all species combined and zooplankton biomass were significant and relatively high (*e.g.*, $0.19 \leq r \leq 0.36, P < 0.05$) in December to February (Table 2).

During the traditionally high discharge months (*i.e.*, March and April) zooplankton biomass increased shelf-wide in waters < 70 m deep (Fig. 1). Small larvae (modal length < 4.0 mm TL) of *Cynoscion arenarius* were most abundant on the inner shelf in April, but larvae abundance correlated significantly with zooplankton biomass over all depth groups (Table 3).

Consequently, these findings are suggestive of additional needed research into the role that plume or CBL waters may play in the food-web dynamics and

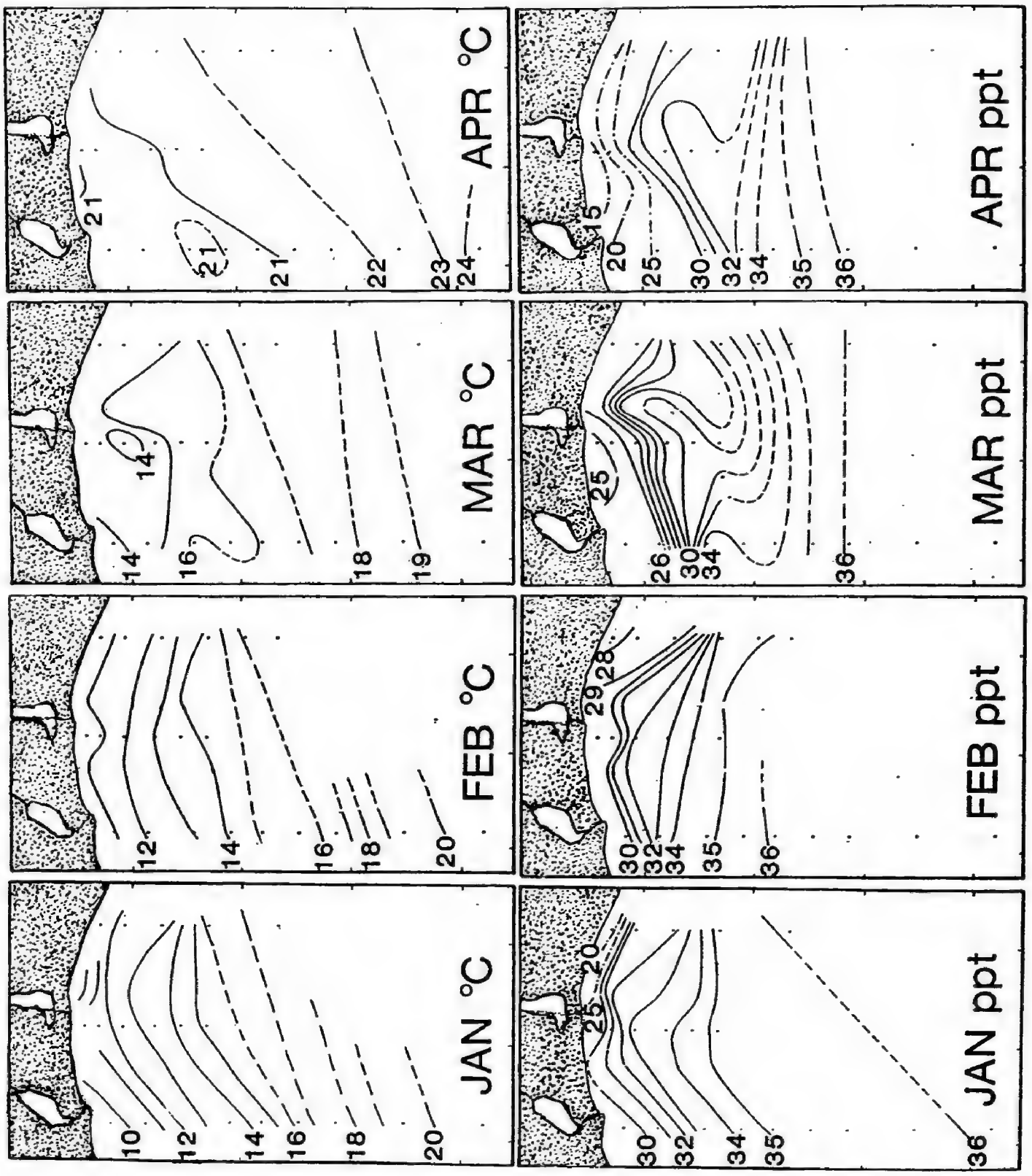


FIG. 2. Surface temperature (° C) and salinity (‰) contours in the northern Gulf of Mexico in January to April 1982 (figure redrawn from Wiseman *et al.* 1986).

resultant survival and recruitment success of fall-winter, shelf-spawned fish larvae even well downstream of the source of the riverine input, *i.e.*, west of the Atchafalaya and Mississippi River deltas (Smetacek 1986; Townsend, *et al.* 1987; Dagg, *et al.* 1988). These correlative results, however, potentially may be a reflection of the interaction of those winter-spawning planktivorous

adult fishes (*i.e.*, *B. patronus* and *E. teres*) and the increased zooplankton biomass associated with the seaward edge of CBL waters on the Louisiana shelf.

ACKNOWLEDGMENTS

We thank E.D. Houde, C. Zastrow, E. Rutherford and J. Watts for their critical review of this manuscript. Funding was provided by a Louisiana Department of Wildlife and Fisheries, U.S. Department of Energy and Louisiana State University Center for Wetland Resources Cooperative Agreement No. DE-FC96-81P6/0313. This is Contribution No. 2138, Center for Environmental and Estuarine Studies of the University of Maryland System.

LITERATURE CITED

AL-YAMANI, F. 1988. Distributional ecology of zooplankton and fish larvae (spot, croaker, menhaden) in the northern Gulf of Mexico. Ph.D. Thesis, University of Miami, Coral Gables, Florida. 420 pp.

ANGELOVIC, J.W. 1976. Plankton and Fisheries Investigations, Vol. 1, 162 pp., 40 figs., basic data in appendices A-F. In NOAA Final Report, Environmental Studies of the South Texas Continental Shelf 1975, U.S. Department of Commerce, National Marine Fisheries Service, Galveston, Texas (Available from the National Technical Information Service, Springfield, Virginia, Report PB283-872AS).

COCHRANE, J.D. and F.J. KELLY, Jr. 1986. Low-frequency circulation on the Texas-Louisiana continental shelf. *Journal of Geophysical Research.* 91:10,645-10,659.

COLTON, J.B., JR., W.G. SMITH, A.W. KENDALL, JR, P.L. BERRIEN and M.L. FAHAY. 1979. Principal spawning areas and times of marine fishes, Cape Sable to Cape Hatteras. *Fishery Bulletin.* **76**:911-915.

COWAN, J.H., Jr. and R.F. SHAW. 1988. The distribution, abundance, and transport of larval sciaenids collected during winter and early spring from the continental shelf waters off west Louisiana. *Fishery Bulletin.* **86**:129-142.

CROUT, R.L. 1983. Wind-driven, near-bottom currents over the west Louisiana inner continental shelf. Ph.D. Thesis, Louisiana State University, Baton Rouge, Louisiana. 117 pp.

DAGG, M.J. 1988. Physical and biological responses to the passage of a winter storm in the coastal and inner shelf waters of the northern Gulf of Mexico. *Continental Shelf Research.* **8**:167-178.

_____, P.B. ORTNER and F. AL-YAMANI. 1988. Winter-time distribution and abundance of copepod nauplii in the northern Gulf of Mexico. *Fishery Bulletin.* **86**:319-330.

DINNEL, S.P. and W.J. WISEMAN. 1986. Fresh water on the Louisiana and Texas shelf. *Continental Shelf Research.* **6**:765-784.

DITTY, J.G., G.G. ZIESKE and R.F. SHAW. 1988. Seasonality and depth distribution of larval fishes in the northern Gulf of Mexico above latitude 26° 00′ N. *Fishery Bulletin.* **86**:811-822.

FAHAY, M.P. 1983. A guide to the early stages of marine fishes occurring in the western Atlantic ocean (with emphasis on species found between Cape Hatteras and the Nova Scotian shelf). U.S. Department of Commerce, NOAA, NMFS, Northeast Fishery Center, Sandy Hook Laboratory. 414 pp.

FINUCANE, J.H., C.B. GRIMES and S.P. NAUGHTON. 1990. Diets of young king mackerel and Spanish mackerel in the U.S. Gulf of Mexico and South Atlantic Bight. *Northeast Gulf Science.* **11**:145-153.

_____, L.A. COLLINS and J.D. McEACHRAN. 1977. Environmental studies of the south Texas outer continental shelf 1976: Ichthyoplankton/mackerel eggs and larvae. NOAA, NMFS,

Galveston, Texas. Report to the Bureau of Land Management, Washington, D.C. PB-283-873, 484 pp.

GOVONI, J.J. and A.J. 1990. Diet composition of larval *Leiostomus xanthurus* in and about the Mississippi River Plume. *Journal of Plankton Research.* 12:819-830.

———, ———, D.E. HOSS and P.B. ORTNER. 1985. An observation of episodic feeding and growth of larval *Leiostomus xanthurus* in the northern Gulf of Mexico. *Journal of Plankton Research.* 7:137-146.

———, D.E. HOSS and D.R. COLBY. 1989. The spatial distribution of larval fishes about the Mississippi River plume. *Limnology and Oceanography.* 34:178-187.

HANSON, R.B. 1982. Influence of the Mississippi River on the spatial distribution of microheterotrophic activity in the Gulf of Mexico. *Contributions in Marine Science.* 25:181-198.

HO, C. and R.B. BARRETT. 1975. Distribution of nutrients in Louisiana's coastal waters influenced by the Mississippi River. Technical Bulletin of the Louisiana Wildlife and Fisheries Commission; Oysters, Wetland Bottoms, Seafood Division, No. 17.

HOUDE, E.D. 1974. Research on eggs and larvae of fishes in the eastern Gulf of Mexico. pp. 187-204. In R.E. Smith (ed.), Proceedings, Marine Environmental Implications of Offshore Drilling in Eastern Gulf of Mexico. State University System of Florida, Institute of Oceanography, St. Petersburg.

ILES, T.D. and M. SINCLAIR. 1982. Atlantic herring: stock discreteness and abundance. *Science.* 215:627-633.

MACKAS, D.L., K.L. DENMAN and M.R. ABBOTT. 1985. Plankton patchiness: biology in the physical vernacular. *Bulletin of Marine Science.* 37:652-674.

SAKAMOTO, W. and Y. TANAKA. 1986. Water temperature patterns and distributions of fish eggs and larvae in the vicinity of a shallow sea front. *Bulletin of the Japanese Society of Scientific Fisheries.* 52:767-776.

SAS INSTITUTE, INC. 1985. SAS user's guide: statistics, Version 5 edition. SAS Institute, Inc. Carry, North Carolina. 956 p.

SAVILLE, A. 1964. Estimation of the abundance of fish stocks from egg and larval surveys. *Rapports et Procès-Verbaux des Reunions de L'Exploration Mer.* 153:164-170.

SHAW, R.F., J.H. COWAN, JR. and T.L. TILLMAN. 1985a. Distribution and abundance of *Brevoortia patronus* (Gulf menhaden) eggs and larvae in the continental shelf waters of west Louisiana. *Bulletin of Marine Science.* 36:96-103.

———, W.J. WISEMAN, JR., R.E. TURNER, L.J. ROUSE, W.-S. CHUANG and F.J. KELLY, Jr. 1985b. The transport of *Brevoortia patronus* (Gulf menhaden) larvae in the continental shelf waters of western Louisiana: a hypothesis. *Transactions of the American Fisheries Society.* 114:452-460.

——— and D.L. DRULLINGER. 1990a. Early life history profiles, seasonal abundance and distribution of four species of clupeid larvae from the northern Gulf of Mexico. NOAA Technical Report, NMFS 88. 60 pp.

——— and ———. 1990b. Early life history profiles, seasonal abundance and distribution of four species of carangid larvae off Louisiana, 1982 and 1983. NOAA Technical Report, NMFS 89. 37 pp.

SMETACEK, V.S. 1986. Impact of freshwater discharge on production and transfer of materials in the marine environment. pp. 85-106. In S. Skreslet (ed.) *The Role of Freshwater Outflow in Coastal Marine Ecosystems.* Springer-Verlag, New York.

SOGARD, S.M., D.E. HOSS, and J.J. GOVONI. 1987. Density and depth distribution of larval Gulf menhaden, *Brevoortia patronus*, Atlantic croaker, *Micropogonias undulatus*, and spot, *Leiostomus xanthurus*, in the northern Gulf of Mexico. *Fishery Bulletin.* 85:601-609.

TEMPLE, R.F., D.L. HARRINGTON and J.A. MARTIN. 1977. Monthly temperature and salinity measurements of continental shelf waters in the western Gulf of Mexico 1963-1965. NOAA Technical Report, SSRF-707. 29 pp.

TOWNSEND, D.W., J.P. CHRISTENSEN, D.K. STEVENSON, J.J. GRAHAM and S.B. CHENOWETH. 1987. The importance of a plume of tidally-mixed water to the biological oceanography of the inner Gulf of Maine. *Journal of Marine Research.* 45:699-728.

WISEMAN, W.J., R.E. TURNER, F.J. KELLY, W.-S. CHUANG, L.J. ROUSE, Jr. and R.F. SHAW. 1986. Analysis of biological and chemical associations near a turbid coastal front during winter 1982. *Contributions in Marine Science.* **29**:141-151.

_____,S.P. MURRAY, J.M. BANE and M.W. TUBMAN. 1982. Temperature and salinity variability within the Louisiana Bight. *Contributions in Marine Science.* **25**:109-120.

WOLANSKI, E. and W.M. HAMNER. 1957. Topographically controlled fronts in the ocean and their biological influence. *Science.* **124**:177-181.

Contributions in Marine Science. (1991) Vol. 32.

OCCURRENCE OF FAT SNOOK (*CENTROPOMUS PARALLELUS*) IN TEXAS: EVIDENCE FOR A RANGE EXTENSION

J.H. Martin and T.L. King[1]

Texas Parks and Wildlife Department
P.O. Box 181078
Corpus Christi, Texas 78480

ABSTRACT

Recent Texas Parks and Wildlife Department investigations have identified a fat snook (*Centropomus parallelus*) population along the extreme south Texas Gulf coast. Identification of fat snook was based on meristic and biochemical characteristics. Sarcoplasmic protein extracts of fat snook and resident congeneric snook (*Centropomus undecimalis*) were compared by thin-layer polyacrylamide gel isoelectric focusing to distinguish protein phenotypes of the two species. It is unknown if these fat snook represent a resident breeding population; however, juvenile fat snook (< 100 mm TL) as well as three females (385, 390, 400 mm TL) in advanced stages of ovum development have been collected.

INTRODUCTION

Snook species of the genus *Centropomus* inhabit tropical and subtropical estuaries of North, Central and South America where they constitute an important recreational and commercial resource (Rivas 1962; Martin and Shipp 1977). Centropomid distribution is at least partially limited by thermal tolerance (Shafland and Foote 1983), freshwater inflow (Fraser 1977) and availability of mangrove stands (Rivas 1986) and grass beds (Gilmore, Donohoe and Cooke 1983). Further, it has been suggested that snook species may be extirpated from portions of their current ranges unless management programs are developed to abate habitat loss and fishing pressure (Rivas 1986).

In the western Gulf of Mexico the northern limit of fat snook (*Centropomus parallelus*) is reported as Tamaulipas, Mexico (Rivas 1986; Robins, Ray and Douglass 1986). Recent Texas Parks and Wildlife Department (TPWD) attempts to collect snook (*Centropomus undecimalis*) for artificial propagation have resulted in the discovery of a putative fat snook population in extreme south Texas.

---

[1] Texas Parks and Wildlife Department, Perry R. Bass Marine Fisheries Research Station, Star Route Box 385, Palacios, Texas 77465.

This report documents the collection and identification of fat snook along the Texas Gulf coast. Fat snook were identified based on meristic characters and verified as biochemically distinct from the congeneric snook utilizing thin-layer isoelectric focusing (IEF) (Lundstrom 1977).

MATERIALS AND METHODS

Nineteen fat snook (80-420 mm TL) were collected October 1986 through September 1989 in the Rio Grande, in the Gulf of Mexico 12 km north of the mouth of the Rio Grande and in the Brownsville Ship Channel 21 km upchannel of Brazos Santiago Pass. Fat snook were collected by rod-and-reel, bag seine and gill net. Live fish were transported to the Texas Parks and Wildlife Department's Perry R. Bass Marine Fisheries Research Station (MFRS), Palacios, Texas and held in culture ponds or tanks. Dead or moribund fish were frozen immediately.

Fat snook (3) and snook (3) were identified to specific level utilizing dichotomous keys of Rivas (1962) and Robins, *et al.* (1986). Diagnostic characteristics included scale size, number of lateral line scales and number of gill rakers on the first gill arch (Robins, *et al.* 1986).

Isoelectric focusing (IEF) was conducted on muscle tissue excised from the lateral musculature immediately below the dorsal fin of thawed specimens. Tissues were homogenized in equal volumes of deionized water and centrifuged at 1,400 g for 15 min. The resultant supernatant was retained for analysis. IEF was performed on 0.25-mm polyacrylamide gels. The gel solution consisted of 2 ml of 29.1% (wt/vol) acrylamide and 0.9% n,N′-methylenebisacrylamide, 0.6 ml of pH 3-10 ampholytes (Serva Fine Biochemicals, Westbury, NY), and 70 µl of TEMED. The gel solution was placed into a flask and degassed for 5 min. Polymerization was initiated by the addition of 70 µl of 10% ammonium persulfate to the gel solution. The acrylamide solution was poured onto a GELBOND PAG film (FMC Bioproducts, Rockland, MN) placed on the lower section of a gel mold. The gel was allowed to polymerize for 1 h before the mold was disassembled and the GELBOND PAG film, with gel attached, was applied to the IEF apparatus.

IEF was performed on an LKB 2117 Multiphor II electrofocusing apparatus powered by a LKB 2197 power supply and thermoelectrically cooled by an LKB 2219 Multitemp II thermostatic circulator (LKB Instruments, Gaithersburg, MD). Electrode strips were soaked in 0.05 M aspartic acid, 0.05 M glutamic acid (anolyte) and 0.05 M arginine, 0.05 M lysine (catholyte) and placed parallel to each other at opposite edges of the gel corresponding to the electrodes on the apparatus. The gel was prefocused for 1 h to ensure the establishment of a consistent pH gradient. The protein extracts were loaded onto a 15 sample application mask (2 µl capacity), and focused for an additional 2 h or until 0 mA was achieved. Focusing was performed at 10° C, 1200 V, 20 mA and 4 W. Gels were fixed for 5 min in a solution of 4% sulfosalicylic acid and 12.5% trichloroacetic acid. Following fixation, gels were placed in 300 ml of wash solution consisting of 40% methanol and 10% glacial acetic acid for 3 minutes. Protein phenotypes were stained with a 5% Coomassie Blue R-250, 40% methanol, and 10% glacial acetic acid solution for 10 minutes. Gels were destained in wash solution until the background cleared and were allowed to air dry. Gels were examined visually. Specimens and polyacrylamide gels are on deposit at MFRS, Palacios, Texas.

RESULTS AND DISCUSSION

Comparisons of meristic and biochemical characteristics confirm the identity of fat snook captured in Texas marine waters. Fish identified as fat snook exhibited more lateral line scales (80) and gill rakers on the first arch (13) than did snook (70-77 scales; 9 gill rakers). Isoelectric focusing of sarcoplasmic protein extracts indicated muscle protein phenotypes of snook

and fat snook differed. The muscle protein electrographs for the two species (Fig. 1) suggest clear protein differences in the lower pH range. Fat snook possess one slightly more acidic band of proteins than do snook. The highly acidic bands are possibly parvalbumin proteins; possessing the unique characteristics of low isoelectric points and low molecular weight (Whitmore 1986). There appears to be additional protein differences in the mid-pH range. However, the pattern is indistinct at the pH gradient employed.

The origin of these fat snook is unknown. Perhaps these are transient individuals from a fat snook population endemic to the Mexican Gulf coast and south Texas represents a seasonal extension of the foraging range. Life history surveys suggest, however, that no extensive migrations occur among any of centropomid species (Volpe 1959, Fraser 1977).

There is evidence that these fat snook represent a resident breeding population. Juvenile fat snook (< 100 mm TL) and three female fat snook (385, 390, 400 mm TL) possessing vitellogenic ova have been collected by TPWD personnel. Water quality and habitat characteristics in the lower Laguna Madre, Texas appear similar to those described as advantageous to

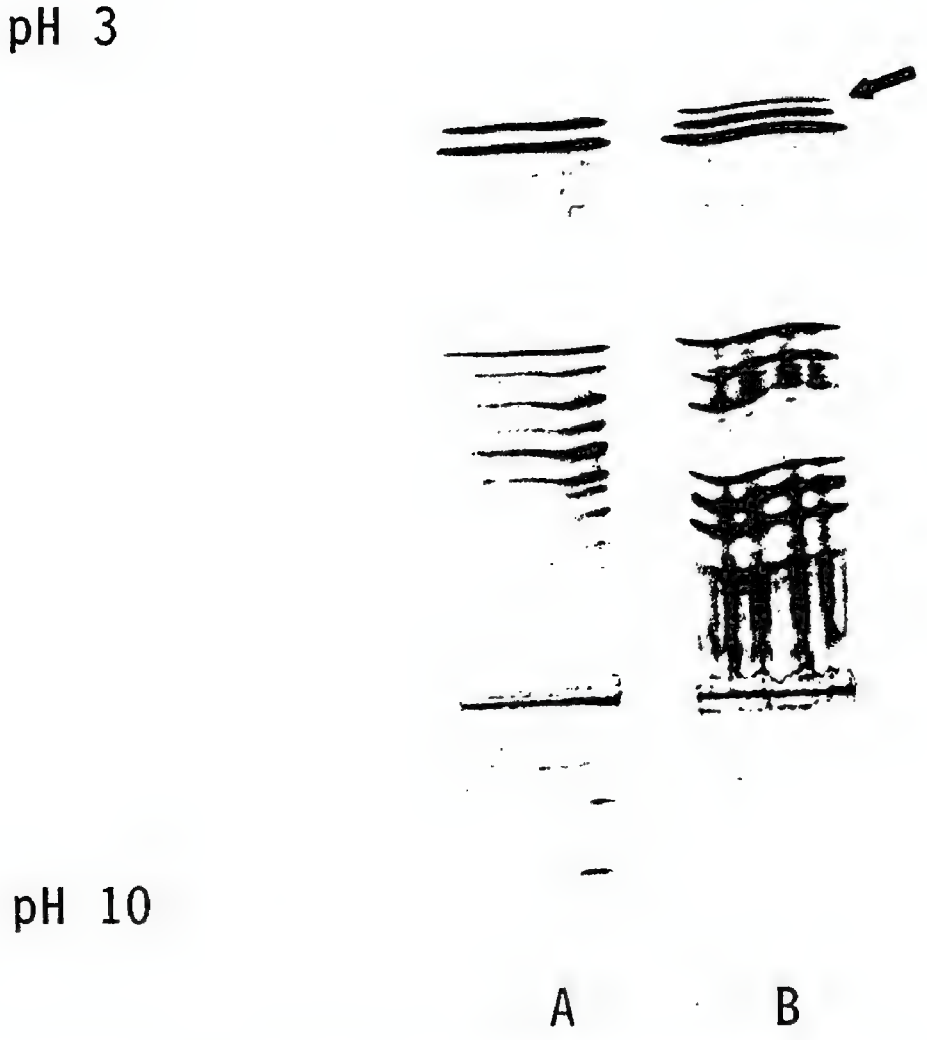


FIG. 1. Electropherograms derived from sarcoplasmic protein extracts of (A) fat snook, *Centropomus parallelus* and (B) snook, *Centropomus undecimalis*. Arrow indicates additional proteins present in fat snook muscle tissue.

growth and reproduction of other centropomid species (Shafland and Foote 1983; Rivas 1986).

The presence of favorable environmental conditions and the occurrence of juvenile and gravid fish combined with the repeated capture of fat snook from the Brownsville Ship Channel and from the Gulf of Mexico up to 12 km north of the Rio Grande suggests that the known range of this species is now extended northward into extreme south Texas.

LITERATURE CITED

FRASER, T.H. 1977. Centropomidae, pp. 1-12. In W. Fisher (ed.), FAO Species Identification Sheets, Fishing Area 31 (West Central Atlantic). FAO, Rome.

GILMORE, R.G., C.J. DONOHOE, and D.W. COOKE. 1983. Observations on the distribution and biology of east-central Florida populations of the common snook, *Centropomus undecimalis* (Bloch). *Florida Scientist.* **46**:306-313.

LUNDSTROM, R.C. 1977. Identification of fish species by thin-layer polyacrylamide gel isoelectric focusing. *Fishery Bulletin.* **75**:571-576.

MARTIN, J.R., and R.L. SHIPP. 1977. Occurrence of juvenile snook, *Centropomus undecimalis*, in North Carolina waters. *Transactions of the American Fisheries Society.* **106**:131-132.

RIVAS, L.R. 1962. The Florida fishes of the genus *Centropomus*, commonly known as snook. *Journal of the Florida Academy of Sciences.* **25**:53-64.

_____. 1986. Systematic review of the perciform fishes of the genus *Centropomus*. *Copeia.* **1986**:579-611.

ROBINS, C.R., G.C. RAY and J. DOUGLASS. 1986. *A Field Guide to Atlantic Coast Fishes of North America.* Houghton Mifflin Company, Boston. 354 pp.

SHAFLAND, P.L., and K.J. FOOTE. 1983. A lower lethal temperature for fingerling snook, *Centropomus undecimalis*. *Northeast Gulf Science.* **6**:175-177.

VOLPE, A. V. 1959. Aspects of the biology of the common snook, *Centropomus undecimalis* of southwest Florida. Florida Board of Conservation Technical Series, No. 31. 37 pp.

WHITMORE, D. H. 1986. Identification of sunfish species by muscle protein isoelectric focusing. *Comparative Biochemistry and Physiology.* **84B**:177-180.

Contributions in Marine Science. (1991) Vol. 32.

SEASONAL VARIATION IN RELATIVE ABUNDANCE AND SPECIES DIVERSITY OF FISHES IN SOUTH BAY

John H. Hook[1]

Biology Department
The University of Texas at Pan American
Edinburg, Texas

ABSTRACT

Seasonal variation in relative abundance and species diversity of fishes in South Bay, Texas was investigated using seine and trawl samples over a 14 month period in 1984 and 1985. Over 115,000 individuals representing 90 species of 35 families were collected. This species count is both larger and substantially different than previously reported from the area. High species richness is due to subtropical location, multiple habitat types and close proximity to a pass into the Gulf of Mexico. Relative abundance varied seasonally between species and within season between collecting sites. Much of the seasonal variation is due to the presence of large numbers of juveniles. Temperature and salinity influenced abundance and diversity on a shorter interval than season.

INTRODUCTION

Fish assemblages of the bays and marshes along the Gulf of Mexico coast have been discussed by many authors over the last 50 years. Most of these investigations concentrated on individual bay or estuarine systems and included species lists with some information on seasonal abundance.

Fish community investigations such as Shenker and Dean (1979); Phillips (1981) and Bell, Pollard, Burchmore, Pease and Middleton (1984) have included relative abundance information but have emphasized species diversity indices absent from earlier studies. Some investigations from northwestern Florida include this information (Subrahmanyam and Drake 1975; Livingston 1976; Subrahmanyam and Coultas 1980) but little comparable data exists from the northwestern Gulf of Mexico. Bechtel and Copeland (1970) provide the only information of this type from Texas.

Although considerable comparative data exist on the species composition of the bays of the middle and northern Texas coast, the lower Laguna Madre remains one of the least investigated bay systems of the northwestern Gulf of Mexico. South Bay at the southern end of the lower Laguna Madre, has received even less investigation. Breuer (1962) had one sampling site in South Bay, but limited the sampling to quarterly trawls. Consequently, his study was

---

[1] Present address: Department of Biology, The University of Southwestern Louisiana, Lafayette, Louisiana 70504.

limited to deeper water habitat. Cain and Dean (1976) pointed out that this sampling method can lead to considerable sampling bias at sites that have large intertidal areas. Because South Bay is a shallow body of water (mean depth = *ca.* 0.5m) with very little shore slope, much of the habitat is intertidal.

The species composition of South Bay is of particular interest in that prior to the dredging of the Brownsville Ship Channel in 1938 it was simply the southern extreme of the Lower Laguna Madre. Placement of spoil isolated South Bay except for a narrow pass connecting it with the Lower Laguna Madre. The effect of this dredging was to decrease mean depth by 50% to 0.5 m and to close a pass with the Gulf of Mexico at the southern end of the bay (Breuer 1962).

The intent of this investigation is to:

1) adequately sample the habitats of South Bay, including the intertidal areas, to provide a complete listing of fishes found in the area and
2) identify the patterns of seasonal variation in the relative abundance and species diversity of fishes in the South Bay.

MATERIALS AND METHODS

Study Area

South Bay, the southernmost bay in Texas, is a small (1,100 ha) bay located near Port Isabel in Cameron County. The center of the bay is less than 7.5 km north of the mouth of the Rio Grande. At its northern end, South Bay opens into the Brownsville Ship Channel approximately 4 km from the channel entrance into the Gulf of Mexico at Brazos Santiago Pass.

Eight sampling sites were established in South Bay along the perimeter and interior of the bay (Fig. 1). Sites were selected to represent as many different habitat types as possible. Seine samples were numbered sequentially according to clockwise position around the bay perimeter beginning at the northern entrance and ending along the western shore. Samples were collected at Sites 1-5 by seine and at Sites 6-8 by trawl. Trawl collections were not used in data analysis except in formulating the species abundance list.

Seine Samples

Samples at Site 1 were collected along the east lip of the entrance channel. Vegetation at all sites varied following a depth gradient. This was most obvious at the edge of the channel at Site 1. *Halodule wrightii* was most abundant at intertidal locations, *Thalassia testudinum* and *Syringodium filiforme* were common either mixed or monospecific beds slightly deeper, and *Halophila engelmannii* occupied the deepest areas where seagrass was present. Site 2 was located in a tidal creek that bisected a stand of black mangrove, *Avicennia germinans*. The creek was approximately 50 m long and 3 m wide. Site 3 was located in the open bay 200 m west of site 2 in an area of mixed seagrass below low tide level. Site 4 samples were taken across near shore zones in an area of comparatively steeper shore slope. The horizontal fluctuation of the shoreline between tides was less than 5 m. In contrast, the same measurement at Site 1 was 175 m. Samples at Site 5 were taken from the shallow, sparsely vegetated flats surrounding oyster beds at the back of the bay. This site was consistently more shallow and turbid than the others.

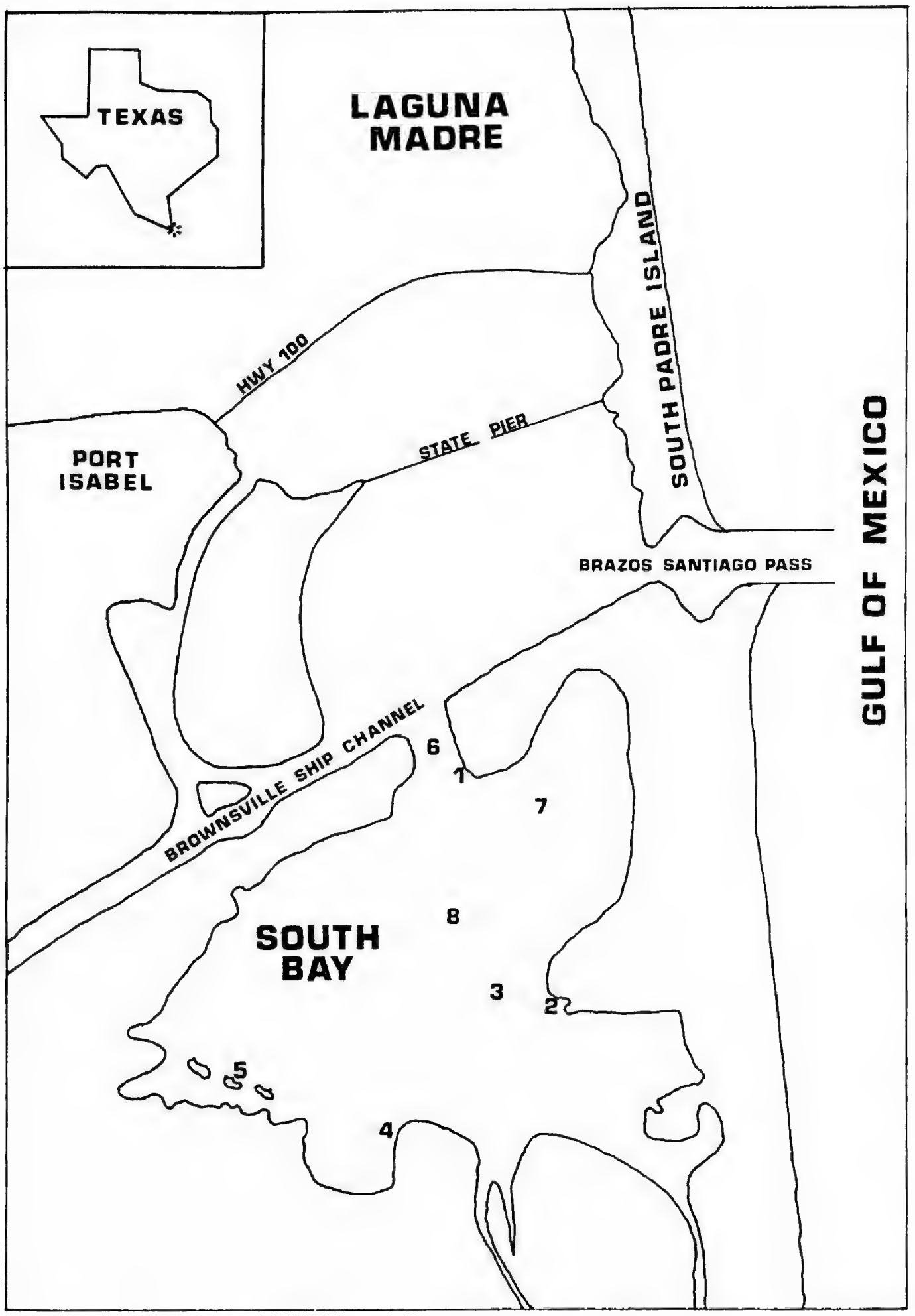


FIG. 1. Map of South Bay. Numbers indicate sampling sites. Sites 1-5 were seine locations, 6-8 were trawl locations.

Trawl Samples

Samples at Site 6 were collected in the deeper water of the entrance channel. Very little vegetation was present in the channel. Mean depth was 3 m. Site 7 samples were collected along the bed of a tributary channel to the entrance channel. Mean water depth was 2 m. For most of its length this channel cut through shallow algal and seagrass flats. There was little gradation between the channel and the flats because the sides of the channel were sharply cut and maintained by strong tidal flow. At high tide, water covering the flats bordering the channel was approximately 1.5 m deep; during an extremely low tide the channel resembled a river with wet grass banks. Site 8 samples were collected in the macroalgal flats located in the eastern portion of the bay. Sample depths were never less than 1.3 or more than 1.5 m.

Collection Techniques and Analysis of Samples

Samples at Sites 1-5 were collected from February, 1984 through March, 1985 at 7 to 14 day intervals. On one occasion samples were taken fewer than 7 days apart and during 2 months (May and November, 1984) only one sample was taken. From February to June, 1984, collections at Sites 2 and 3 were taken alternately, depending upon water level. During high tides samples were collected at Site 2 and at low tide at Site 3. Beginning in July, collections were taken at both sites regardless of water level. Trawl samples at Sites 6-8 were taken during the same time frame but at quarterly intervals. Several species collected in additional trawl samples of South Bay were included in the species composition list.

Collections at Sites 1-5 were made with a 10 m bag seine (8 mm wing mesh, 4 mm bag mesh) and at sites 6-8 with a 4.9 m otter trawl (25 mm cod end mesh). All sample hauls were timed so that catch per unit effort could be calculated. Seine samples were of different duration due to site differences, but trawl samples were all 10 minutes.

At each site mid-water temperature was taken to the nearest 1.0° C for each sample. Salinity data were taken only from June, 1984 through March, 1985. Water samples were returned to the laboratory and titrated for total chloride content following the methods of Cox (1972).

Fish collected were preserved in a 10% formalin solution in the field and returned to the laboratory for identification, weight and measurement. Several individuals, too large for the preservative containers, were placed on ice and analyzed immediately on arrival to the laboratory. Collections containing representative samples and individuals of species uncommon in South Bay have been deposited at the Pan American University Coastal Studies Laboratory at South Padre Island, Texas.

Data Analysis

Seasonal comparisons of relative abundance used catch per unit effort to eliminate bias due to variation in collection times. Thus, the number of individuals was divided by the minutes spent collecting to yield the number of individuals captured per minute.

Three diversity indices were calculated for all samples. Shannon's diversity index ($H' = -\Sigma P_i \ln P_{i,}$ where P_i = the proportion of individuals of the i^{th} species), which increases as number of species and evenness of distribution of abundance among species increases; Margalef's species richness index ($D = (S-1)/\ln N$, where S = the number of species in the sample, and N = the total number of individuals); and Pielou's evenness index ($J = H'/\ln S$) where a value of 1 equals unity was calculated for each sample based on number of individuals. Shannon's index was chosen since it is frequently used, facilitating comparison with other studies. Margalef's and Pielou's indices were chosen to point out the contribution of species richness and relative species abundance in species diversity.

RESULTS

Physical Parameters

Seasonal variation in temperature and salinity followed predictable trends but did have sharp episodic departures from those trends. Water temperatures in the bay ranged from 5 to 39° C, with the maximum occurring during July and the minimum in January. Water temperature dropped from 27 to 5° C at Site 5 following the passage of a cold front during January, 1985.

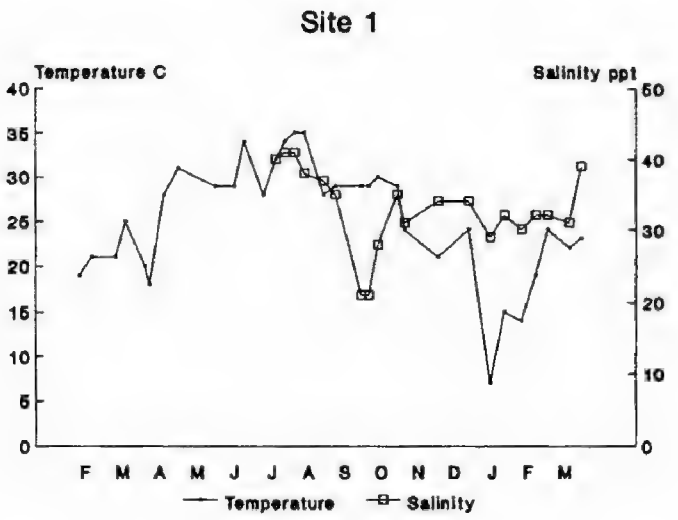
Salinity levels generally were slightly above seawater but did vary between sites although not significantly ($F = 0.5$). Variation in salinity among sites increased along a gradient from the mouth to the back of the bay. Salinity dropped from 37 to 9 ‰ at Site 5 following a heavy rain in September. Salinity remained low for approximately two weeks. All sites had similar seasonal patterns in salinity and temperature with Sites 1 and 5 exhibiting the extremes (Fig. 2).

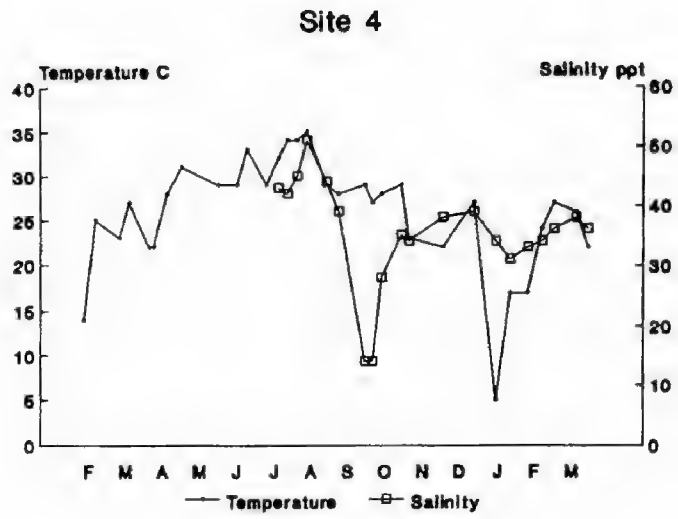
Relative Abundance

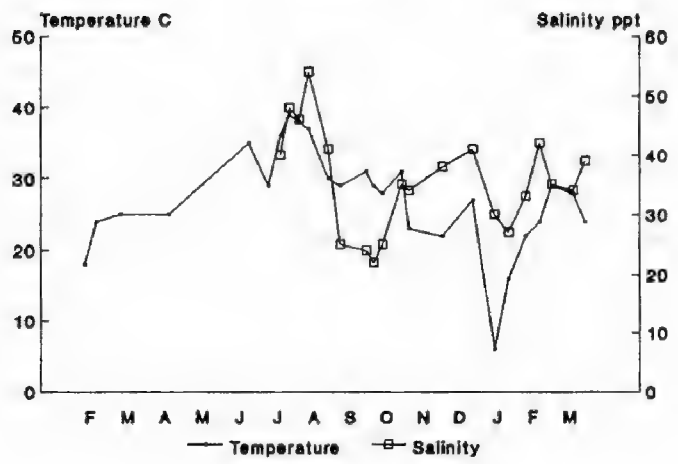
A total of 115,674 individuals of 90 species and 35 families were collected by seine and trawl (Table 1). Two families, Cyprinodontidae and Sciaenidae, represented 45% of the total number of individuals transformed to catch per unit effort (CPUE) and 55% of the total biomass (CPUE) collected. During the study, the most numerous species in the bay was *Leiostomus xanthurus*. *Cyprinodon variegatus* had the greatest biomass. However, these species were not the most abundant during each season. In winter (Dec., Jan., Feb.), *Brevoortia patronus* was most numerous and *Cyprinodon variegatus* had the greatest biomass. In all other seasons the species that was most numerous also had the greatest biomass: *Leiostomus xanthurus* (Mar., Apr., May), *Cyprinodon variegatus* (Jun., Jul., Aug.), *Anchoa mitchilli* (Sep., Oct., Nov.). Seasonal variation among and between the 16 most numerous species (representing 95% of the total individuals collected) is shown in Table 2.

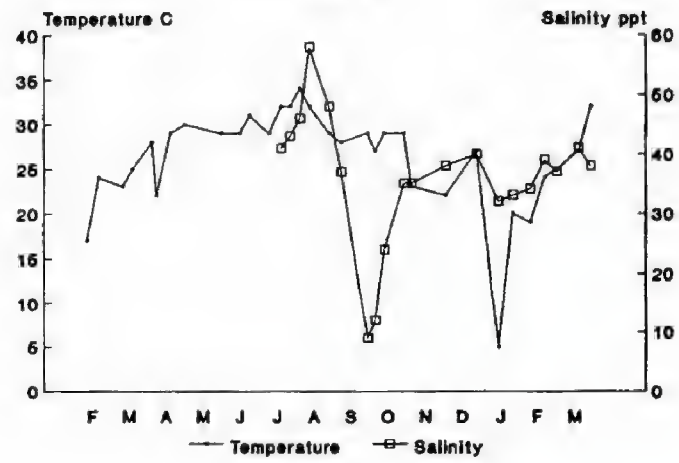
Considerable variation occurred between sites in terms of species relative abundance. The most abundant species was seldom the same in all five seining stations during any one month. At all sites except Site 4 the most numerous species also had the greatest biomass. Over the 14 month study the most abundant species were: Site 1, *Leiostomus xanthurus*; Site 2, *Cyprinodon variegatus*; Site 3, *Menidia peninsulae*; Site 4, *Menidia peninsulae* (individuals); *Lagodon rhomboides* (biomass); Site 5, *Mugil cephalus*.

There were considerable seasonal differences in species composition and dominance between sites. Site 2 was dominated by cyprinodontid species during the summer and winter, and by the atherinid, *Menidia peninsulae*, in the fall. No other site exhibited this pattern. Sites 1, 3, 4 and 5 showed a common trend of dominance by *Leiostomus xanthurus* during February, March and April of 1984, then dominance by *Anchoa mitchilli* during October









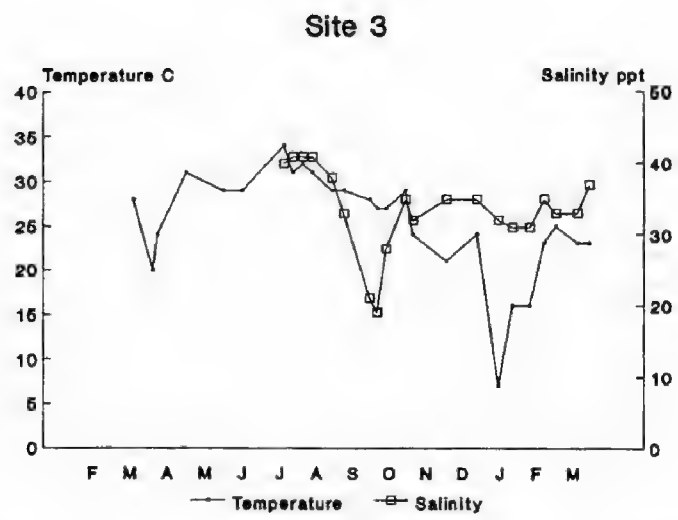


FIG. 2. Seasonal variation in salinity and temperature at Sites 1-5.

TABLE 1
Species listing and total abundance of individuals collected by seine and trawl in south Bay. Species marked (*) were collected only by trawl.

SPECIES	ABUNDANCE	SPECIES	ABUNDANCE
Leiostomus xanthurus	29665	*Bathygobius soporator*	9
Cyprinodon variegatus	22997	*Archosargus probatocephalus*	9
Anchoa mitchilli	14741	*Eucinostomus melanopterus*	8
Menidia peninsulae	12532	*Hemiramphus brasiliensis*	7
Brevoortia patronus	9045	*Opsanus beta*	5
Mugil cephalus	5631	*Caranx hippos*	5
Gobionellus boleosoma	3690	*Gobionellus hastatus*	4
Fundulus similis	3170	*Prionotus tribulus*	4
Lagodon rhomboides	2778	*Sphoeroides parvus*	4
Mugil curema	2251	*Centropomus undecimalis*	3
Anchoa hepsetus	1871	*Trachinocephalus myops*	3
Eucinostomus argenteus	1094	*Serranus subligarius*	3
Sardinella anchovia	905	*Hemicaranx amblyrhynchus*	3
Sciaenops ocellatus	743	*Conodon nobilis*	3
Citharichthys spilopterus	646	*Pogonias cromis*	3
Fundulus grandis	601	*Stellifer lanceolatus*	3
Harengula jaguana	511	*Gobiomorus dormitor*	3
Membras martinica	417	*Symphurus plagiusa*	3
Syngnathus scovelli	388	*Dasyatis sabina* *	2
Micropogonias undulatus	367	*Myrophis punctatus*	2
Bairdiella chrysura	351	*Trachinotus carolinus*	2
Anchoa nasuta	174	*Histrio histrio*	2
Evorthodus lyricus	115	*Diapterus olisthostomus* *	2
Lucania parva	90	*Pomadasys crocro*	2
Orthopristis chrysoptera	90	*Umbrina coroides* *	2
Gobiosoma robustum	72	*Menticirrhus americanus*	2
Gerres cinereus	67	*Monacanthus hispidus*	2
Dormitator maculatus	62	*Trachinotus falcatus*	1
Syngnathus louisianae	60	*Anchoviella perfasciata*	1
Gobiosoma bosci	56	*Hyporhamphus unifasciatus*	1
Polydactylus octonemus	51	*Mycteroperca rubra*	1
Paralichthys lethostigma	46	*Centropristis philadelphica* *	1
Arius felis	41	*Lutjanus griseus*	1
Brevoortia gunteri	36	*Selene vomer* *	1
Cynoscion nebulosus	33	*Chloroscomhrus chrysurus*	1
Hippocampus zosterae	32	*Lutjanus synagris*	1
Elops saurus	26	*Larimus fasciatus* *	1
Adinia xenica	25	*Astroscopus y-graecum*	1
Synodus foetens	15	*Microgobius gulosus*	1
Platybelone argalus	14	*Trichiurus lepturus*	1
Cynoscion arenarius	14	*Scorpaena plumieri* *	1
Strongylura marina	11	*Ancylopsetta quadrocellata* *	1
Eucinostomus gula	11	*Achirus lineatus*	1
Chilomycterus schoepfi	11	*Aluterus schoepfi* *	1
Oligoplites saurus	10	*Lagocephalus laevigatus* *	1

TABLE 2
Monthly variation among the 16 species representing 95% of total abundance, pooled from sites 1-5.

Species	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar
L. xanthurus	327.2	480.2	116.7	21.8	6.6	2.4	2.0	0.3	0.3	0.2	0.5	2.4	9.1	19.8
C. variegatus	37.6	37.7	16.9	0.8	86.0	13.9	212.4	1.0	3.7	4.8	44.7	125.4	46.4	19.3
A. mitchilli	2.3	0.0	0.0	0.7	0.0	0.1	67.7	100.1	80.5	98.7	16.7	1.8	5.4	4.9
M. peninsulae	2.2	12.4	10.0	156.5	65.1	12.4	22.7	8.1	33.1	9.9	48.9	19.1	39.0	8.4
B. patronus	8.6	38.1	36.5	3.6	0.1	0.0	0.0	0.0	0.0	0.0	7.1	4.8	87.1	54.2
M. cephalus	9.6	27.9	27.3	1.6	0.1	1.1	0.6	1.6	1.0	0.0	20.1	33.9	28.0	9.3
G. boleosoma	0.0	0.6	40.0	5.3	9.5	17.7	1.8	1.0	12.5	1.7	0.7	0.1	0.0	3.3
F. similis	1.6	0.4	0.9	0.3	40.7	6.4	25.8	0.1	0.7	0.0	9.8	0.2	0.4	2.5
L. rhomboides	1.5	4.1	15.9	9.1	6.2	5.4	9.2	1.1	1.8	1.4	0.6	1.5	1.0	6.3
M. curema	0.0	0.0	0.0	0.0	88.9	4.3	3.9	0.3	1.2	0.0	0.0	0.0	0.0	0.0
A. hepsetus	0.0	0.0	0.0	0.0	0.0	0.4	18.0	3.5	22.8	0.8	0.2	0.0	0.0	0.0
E. argenteus	0.0	0.0	0.0	0.3	2.8	5.4	8.0	2.3	8.4	1.8	0.0	0.0	0.0	0.0
S. anchovia	0.0	0.0	0.0	0.0	0.1	0.0	20.3	4.5	0.1	0.0	0.0	0.0	0.0	0.0
S. ocellata	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	5.8	4.2	10.6	0.4	0.3	0.3
C. spilopterus	3.6	2.2	12.0	0.0	0.0	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.3
F. grandis	0.0	0.1	0.3	0.1	8.5	2.5	0.8	0.1	1.4	0.0	0.9	4.6	1.2	0.3

and November, followed by *Brevoortia patronus* in March, 1985. Between these periods of dominance by a common species, each site showed a different relative abundance pattern.

The total number of individuals reached peak values in January and February at each site. This was largely due to recruitment of juvenile *Leiostomus xanthurus* into the bay population. Biomass peaked in April and May. Individual and biomass levels decreased markedly during major changes in salinity and temperature at some sites.

Species Diversity

Species diversity (H′) varied greatly with no apparent annual cycles (Fig. 3). However, rough seasonal trends in diversity (H′) and richness (D) were apparent. Mean diversity and species richness values were greater from July through September. Evenness values, however, did not show a seasonal trend. Evenness was inversely correlated ($P < 0.01$) to numerical abundance at all sites (Site 1, r = −0.48; Site 2, r = −0.45; Site 3, r = −0.73; Site 4, r = −0.57; Site 5, r = −0.46). Shannon's H′ ranged from 0.14 to 3.23. Species richness (Margalef's D) ranged from 0.22 to 3.37. Evenness values ranged from 0.03 to 0.97. All three indices decreased in the same pattern following major changes in salinity or temperature. This is shown in Figure 3 using diversity (H′). The other sites did not exhibit the same pattern.

DISCUSSION

Diversity

Evenness fluctuated over the same interval, but inversely with abundance. Because species richness and the distribution of individuals among species are important in determining H′, the short term variation in evenness disrupted the seasonal pattern exhibited by species richness. Thus, H′ showed a rough seasonal trend at each site. This seasonal trend is more clearly seen when site data is pooled (Fig. 4). Pooling moderates site extremes so that diversity trends in the bay become more apparent. Species richness (D) exhibits strong seasonal trends, evenness (J) does not.

Between site variation in species diversity, species richness and evenness may be due in part to habitat variation between sites. Sites 1 and 4 had the highest mean values on all three diversity indices and also had a higher degree of structural variety and complexity than the other sites. Site 2 also had a high degree of vertical structure but lacked the complexity contributed by submergent vegetation. Diversity indices did not vary with changes in salinity and temperature except at Site 3. All measures of diversity decreased markedly with those changes at Site 3. This may be due to a lack of a strong intertidal component in the fish assemblage at Site 3. Cyprinidontid species, which comprise a large percentage of the assemblage in intertidal areas,

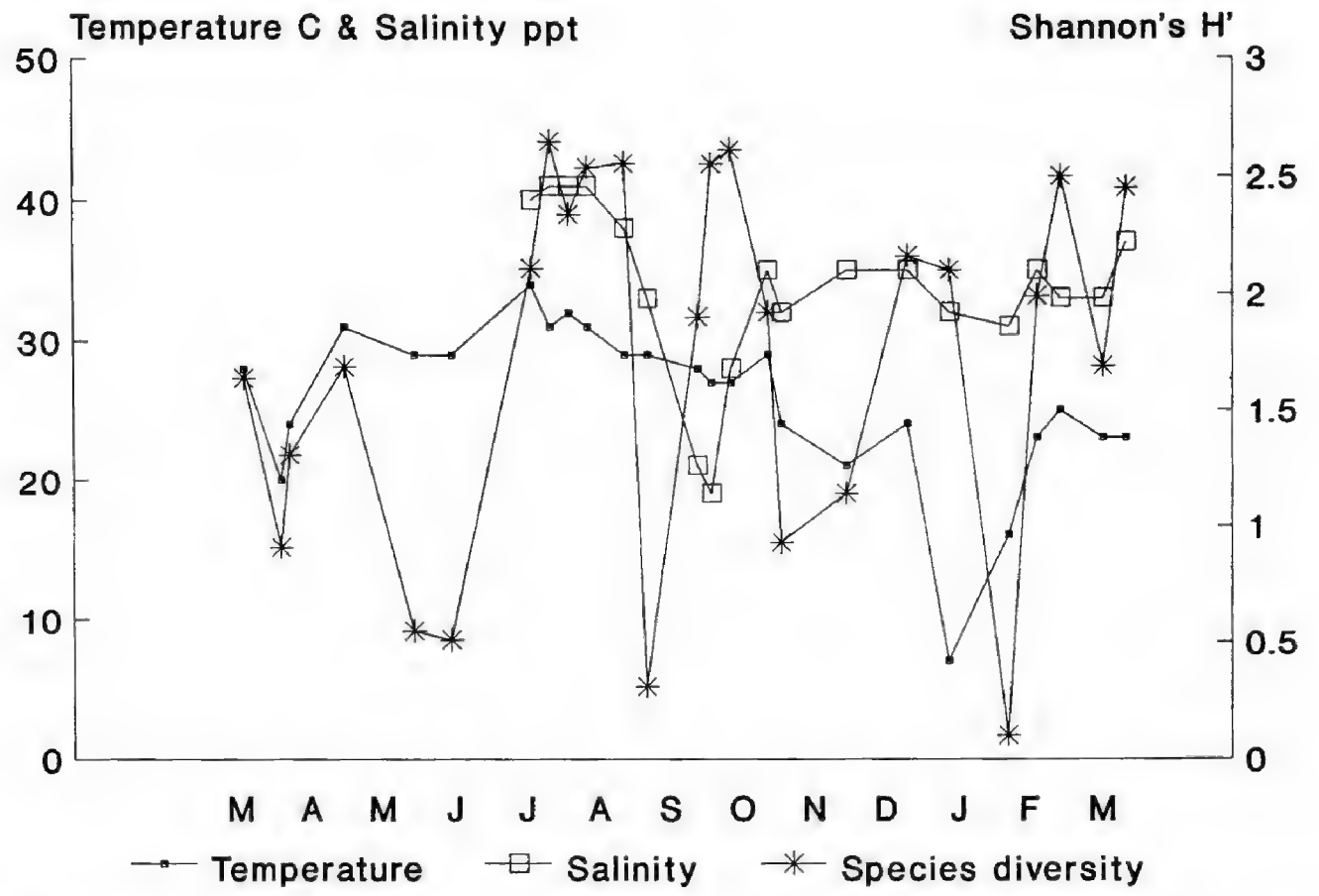


FIG. 3. Species diversity (H') patterns compared to seasonal variation in salinity and temperature at Site 3.

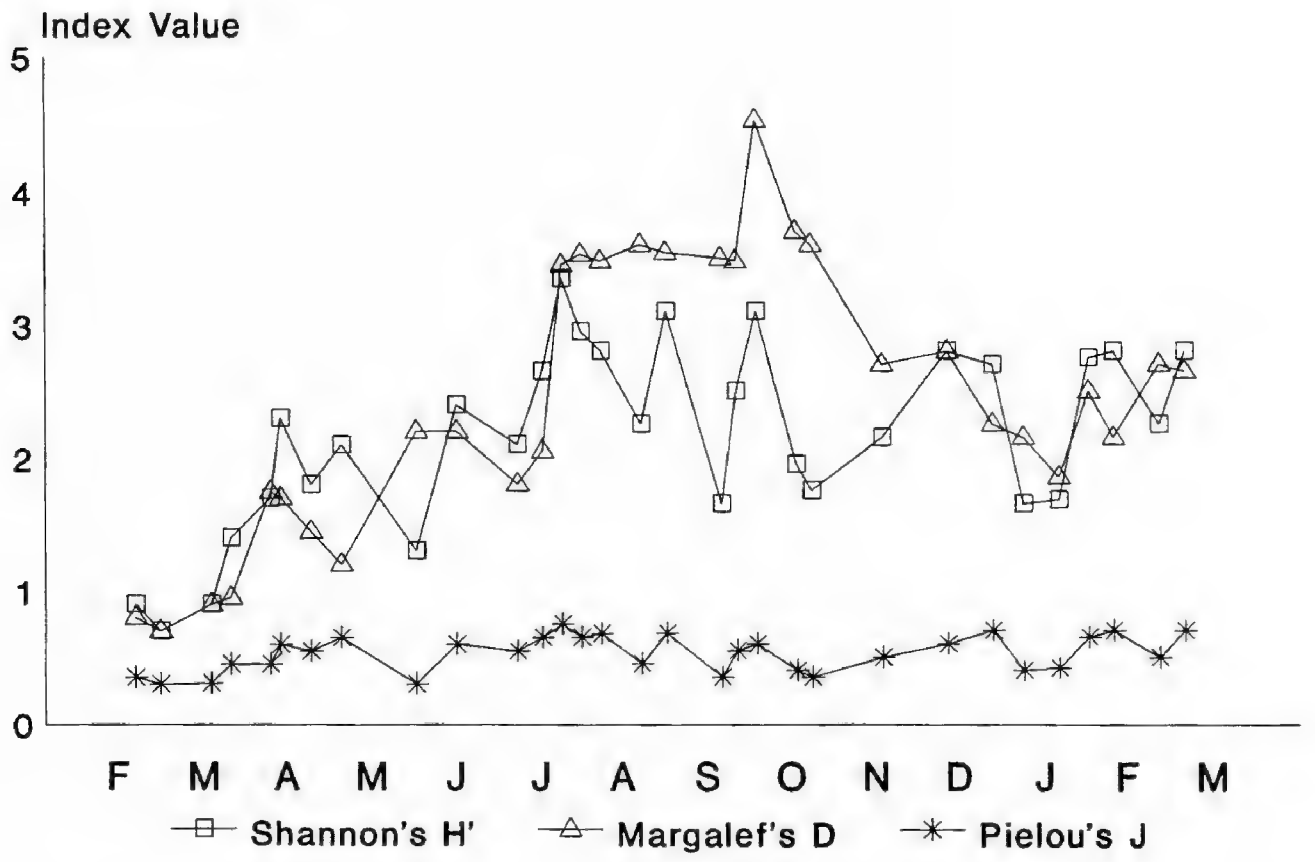


FIG. 4. Seasonal variation in species diversity (H'), species richness (D) and evenness (J). Data pooled from Sites 1-5.

typically exhibit tolerance for wide ranges in environmental conditions. Thus, intertidal areas should see less variation in diversity indices during extremes in environmental conditions than subtidal areas where cypridontid species are not usually abundant.

Species richness and diversity (H′) in South Bay are relatively high when compared to some other Atlantic and Gulf bays and estuaries (Table 3). Several factors contribute to the relatively high number of species found in South Bay. Due to its location at 26° N latitude, South Bay contains species that are normally associated with tropical waters such as *Pomadasys crocro*, *Platybelone argalus* and *Umbrina corroides*. South Bay also includes *Adinia xenica*, a common species of the northern and eastern Gulf of Mexico bay systems that has previously not been reported south of Corpus Christi, Texas. Habitat diversity in South Bay also influences species richness. Oyster clumps and beds, mixed seagrass beds, mixed algal flats, tidal channels and mangrove stands form a complex assortment of habitats in a small area. This allows species strongly associated with particular habitat types, such as *Gobiosoma bosci* and *G. robustum* (Hoese 1966), to occur in the bay.

Faunal composition varied markedly between sites, possibly due to gross differences in habitat. Site 2, a tidal creek with very little vegetation, was dominated by cyprinodontid species during most the year. Species common in

TABLE 3
Species diversity (H′) and species richness as total number of species collected from Atlantic and Gulf of Mexico inshore systems.

Location	Author	Diversity (H') Range	Species (s)
Apalachicola Bay Florida	Livingston (1976)	0.17-1.91	76
Sapelo Sound Georgia	Dahlberg and Odum (1970)	0.70-1.80	70
North Inlet South Carolina	Cain and Dean (1976)	0.42-2.30	51
Galveston Bay Texas	Bechtel and Copeland (1970)	0.90-1.60	66
South Bay Texas		0.14-3.23	90

other parts of the bay such as *Lagodon rhomboides* and *Syngnathus scovelli* were much less abundant at Site 2. Faunal composition at Site 3, a heavily vegetated area offshore and adjacent to Site 2, was very different from that of Site 2. Species associated with open waters, such as *Anchoa mitchilli* and *Brevoortia patronus*, were seasonally dominant while cyprinodontids were rare. Numerous species that were rare in South Bay were collected along the channel edge at Site 1. Vertical substrate, comparatively stable salinity and a distinct temperature gradient of 4 to 7° C along the slope of the channel lip during most collections probably explains increased species richness at Site 1.

The proximity of the bay to Brazos Santiago Pass probably contributes most to the overall species richness in South Bay. Prevailing currents bring water in the pass, through the Brownsville Ship Channel and into the mouth of South Bay. Thus, with every incoming tide South Bay receives fresh seawater from the Gulf of Mexico. This circulation pattern brings in many of the rare species, such as *Centropristis philadelphica*. The periodic input of fresh seawater also moderates salinity levels relative to other areas of the lower Laguna Madre. Since hypersaline conditions depress species richness (Gunter 1967) this continual flushing is probably important in maintaining high species richness in South Bay.

Abundance

Seining shallow, intertidal areas demonstrated the abundance of small fishes in the bay and that some very abundant species are not adequately surveyed by trawling. For example, 28 *Leiostomus xanthurus* were taken in trawls and 29,637 by seine. Although *Leiostomus xanthurus* was the most numerous species in the bay it would have been largely overlooked using only trawl collections. This also probably explains the rather large differences between Breuer's (1962) survey of the Lower Laguna Madre and this investigation. Forty two species reported here were not listed in that report, most of which inhabit shallow or intertidal areas.

Seasonal variation in relative abundance in South Bay is dependant upon spawning and recruitment times. Each species reaches its maximum numeric abundance when juveniles are recruiting into the population. The most abundant permanent residents are serial spawners. Since their spawning times are dependent upon site conditions such as water level, temperature and salinity, their recruitment and abundance are less predictable than the temporary residents. Dominance in the community alternates between seasonal and serial spawners. During months that seasonal spawners make up less than 50% of the total abundance (May, June, July, August, December and January) serial spawners exhibit their seasonal peaks (*C. variegatus* 75%, *M. peninsulae* 62%, *F. similis* 90%, *F. grandis* 78%). This could indicate that the abundance of seasonal spawners partially controls the spawning and

recruitment times for resident serial spawners. Although *G. boleosoma* does not follow this pattern the benthic nature of the species may reduce competition with the dominant species in the water column.

Abundance patterns of two important sciaenids, *Leiostomus xanthurus* and *Sciaenops ocellatus,* indicate that temperature may be an important factor influencing relative abundance during a shorter interval than a season. *L. xanthurus* comprised almost 80% of the individuals collected during February and March 1984, during recruitment of juveniles. Immediately prior and throughout recruitment, water temperatures remained moderate with no sudden declines. The following year, a cold front passed through the area on January 12, 1985, dropping water temperature rapidly to a low of 5° C. *L. xanthurus* dropped to less than 20% of individuals collected during February and March 1985 (Fig. 5). Dawson (1958) observed fish kills including *L. xanthurus* in South Carolina ponds when water temperature neared 5° C, so it would be reasonable to consider interannual variation in relative abundance as due to lethal low temperature.

Sciaenops ocellatus spawns earlier in the fall and is most abundant as juveniles in Texas bays from January to April (Gunter 1945). This recruitment pattern makes observation of low temperature effects on juveniles of a single year class possible. Catch rate for *Sciaenops ocellatus* during fall 1984 through spring 1985 (Fig. 6) declined immediately after the cold front arrived and did not recover later that spring. If cold temperature forced the fish to leave the nursery ground then they should have returned with warmer temperatures. It should be noted that in December, 1983 South Bay was hit by one of the most severe freezes on record. This episode occurred while young juveniles of *S. ocellatus* were present in the bay but prior to arrival of *L. xanthurus* juveniles. February and March, 1984 samples show large numbers of *L. xanthurus* but only a single *S. ocellatus*.

Little accurate information is available on the ecology of South Bay prior to the dredging of the Brownsville Ship Channel. However, there is little doubt that placement of spoil along South Bay following that dredging has had a major impact on the whole bay environment. Restricted water flow at the north end of the bay allowed Boca Chica Pass at the southern end to close. That closure and siltation from unprotected spoil are probably responsible for mean depth reduction from 1 m to 0.5 m following dredging. This reduction created greater area suitable for seagrass beds and tidal channels that did not exist previously. Present conditions in South Bay provide nursery grounds for juveniles of many species of marine fishes as well as habitat for adult fish of species normally associated with bay environments.

ACKNOWLEDGMENTS

Portions of these data were collected to partially fulfill M.S. degree requirements at the University of Texas at Pan American. Robert Edwards was very helpful in preparing an early draft

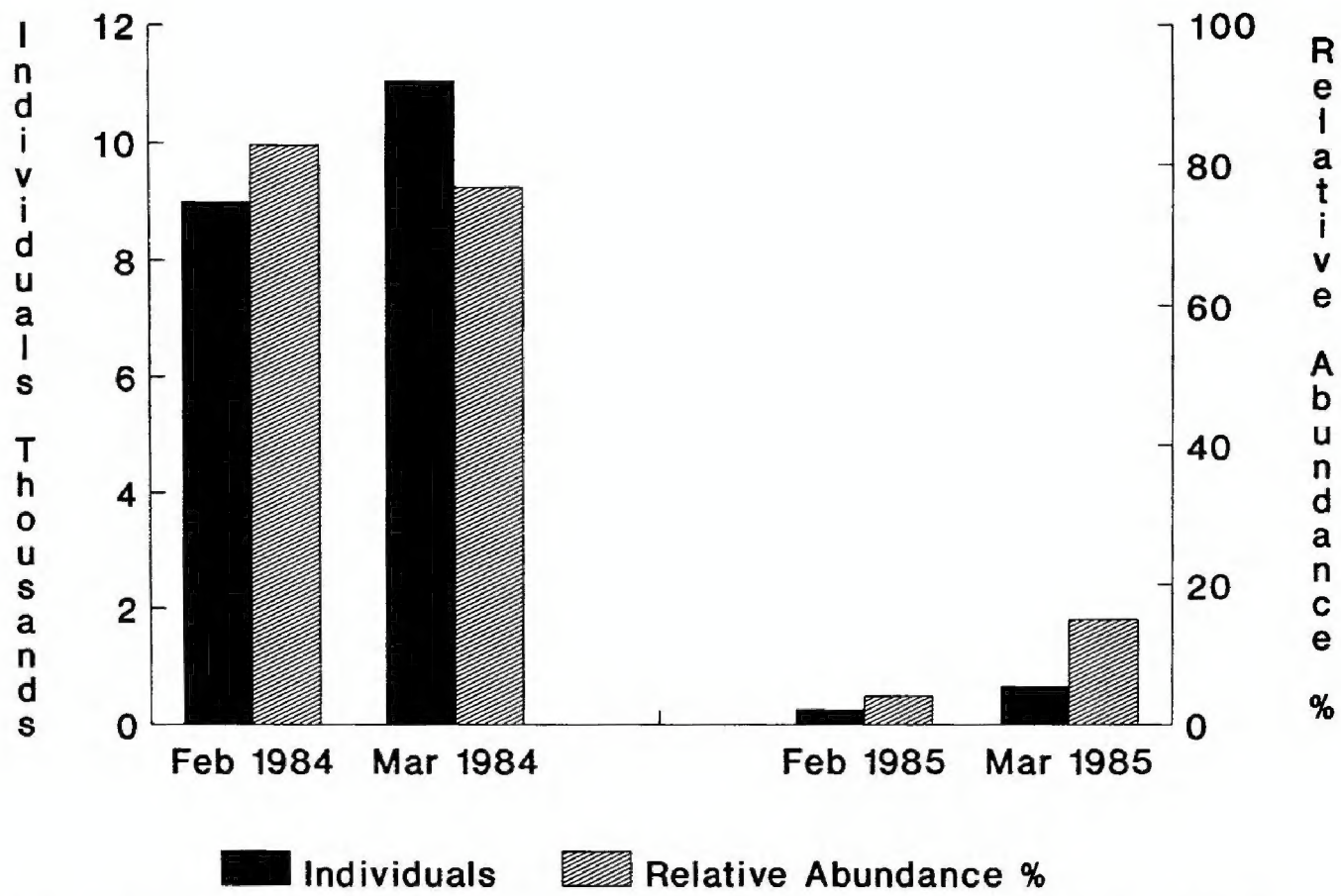


FIG. 5. Abundance of juvenile *Leistomus xanthurus* during recruitment in 1984 and 1985.

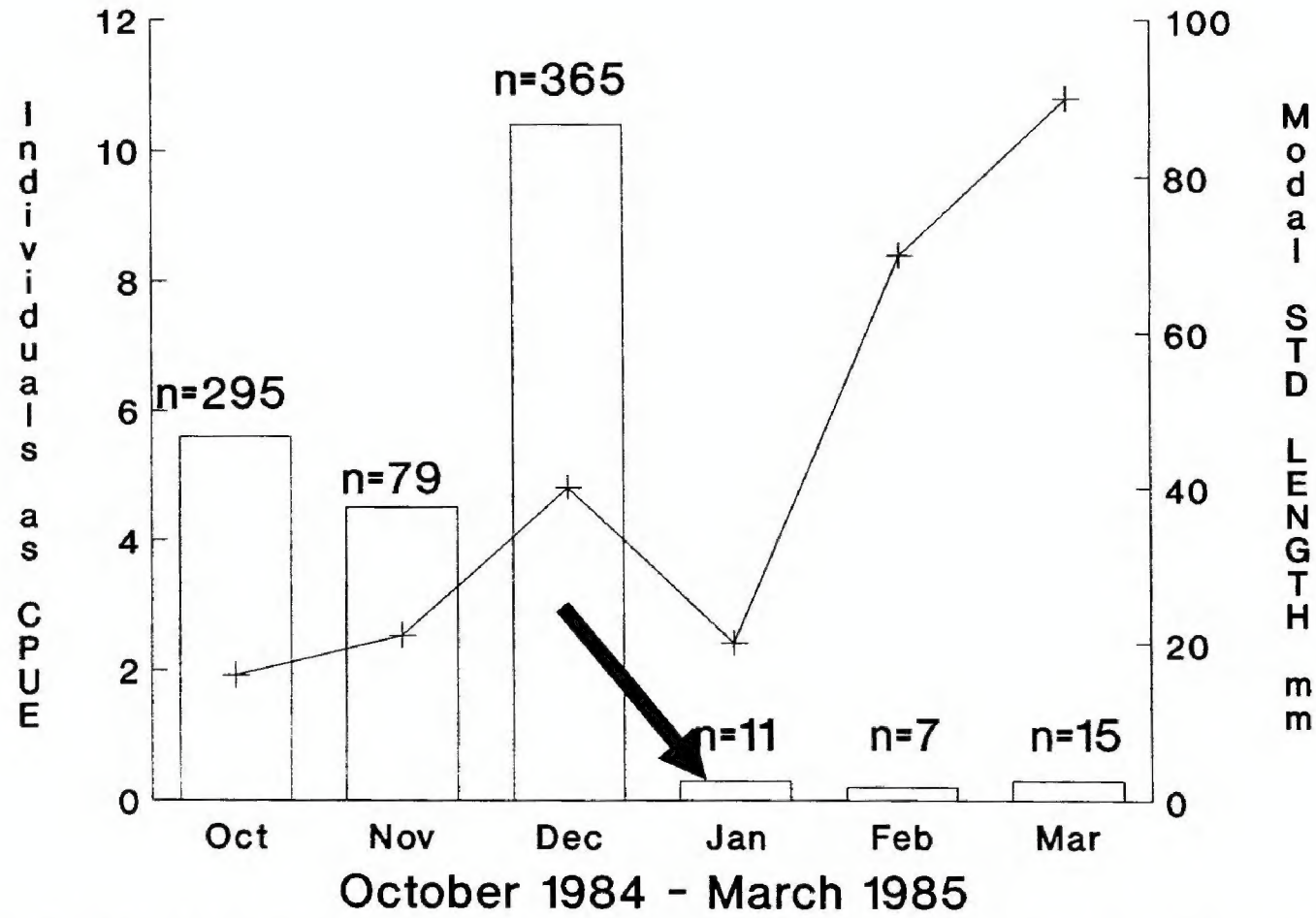


FIG. 6. Abundance (bars) and modal standard length (line) of juvenile *Sciaenops ocellatus* prior to and following a January 1985 "freeze" (temperature dropped from 25 to 4° C at arrow).

of this manuscript. I thank Frank Judd for providing facilities at U.T.P.A. Coastal Studies Laboratory. I also thank H.D. Hoese and an anonymous reviewer for critical manuscript review.

LITERATURE CITED

BECHTEL, T.J. and B.J. COPELAND. 1970. Fish species diversity indices as indicators of pollution in Galveston Bay, Texas. *Contributions in Marine Science.* **15**:103-132.

BELL, J.D., D.A. POLLARD, J.J. BURCHMORE, B.C. PEASE and M.J. MIDDLETON. 1984. Structure of a fish community in a temperate tidal mangrove creek in Botany Bay, New South Wales. *Australian Journal of Marine and Freshwater Research.* **35**:33-46.

BREUER, J.P. 1962. An ecological survey of the lower Laguna Madre of Texas, 1953-1959. *Publications of the Institute of Marine Science, The University of Texas.* **8**:153-183.

CAIN, R.L. and J.M. DEAN. 1976. Annual occurrence, abundance and diversity of fish in a South Carolina intertidal creek. *Marine Biology.* **36**:369-379.

COX, G.W. 1972. *Laboratory Manual of General Ecology.* Wm. C. Brown Company. Dubuque, Iowa.

DAWSON, C.E., JR. 1958. A study of the biology and life history of the spot, *Leiostomus xanthurus* Lacepede, with special reference to South Carolina. *Contributions of the Bears Bluff Laboratory,* **28**. 48 pp.

GUNTER, GORDON. 1945. Studies on marine fishes of Texas. *Publications of the Institute of Marine Science, University of Texas.* **1(1)**:1-190.

———. 1967. Vertebrates in hypersaline waters. *Contributions in Marine Science.* **12**:230-241.

HOESE, H.D. 1966. Habitat segregation in aquaria between two sympatric species of *Gobiosoma. Publications of the Institute of Marine Science, The University of Texas.* **11**:7-11.

LIVINGSTON, R.J. 1976. Diurnal and seasonal fluctuations of organisms in a north Florida estuary. *Estuarine and Coastal Marine Science.* **4**:373-400.

PHILLIPS, P.C. 1981. Diversity and fish community structure in a Central American mangrove embayment. *Revista de Biologia Tropical.* **29**:227-236.

SHENKER, J.M. and J.M. DEAN. 1979. The utilization of an intertidal salt marsh creek by larval and juvenile fishes: abundance, diversity and temporal variation. *Estuaries.* **2**:154-163.

SUBRAHMANYAM, C.H. and S.H. DRAKE. 1975. Studies on the animal communities in two north Florida salt marshes. Part 1. Fish communities. *Bulletin of Marine Science.* **25**:445-465.

——— and C.L. COULTAS. 1980. Studies on the animal communities in two north Florida salt marshes. Part III. Seasonal fluctuation of fish and macoinvertebrates. *Bulletin of Marine Science.* **30**:790-818.